Report of the work of the expert group on maintaining the ability of biodiversity to continue to support the water cycle

Mike Acreman, Juliana Albertengo, Telmo Amado, Mao Amis, Aileen Anderson, Isam Bacchur, Gottlieb Basch, Ademir Calegari, Nick Chappell, Nakul Chettri, et al.

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REPORT OF THE WORK OF THE EXPERT GROUP ON MAINTAINING THE ABILITY OF BIODIVERSITY TO CONTINUE TO SUPPORT THE WATER CYCLE

Note by the Executive Secretary

1. The tenth meeting of the Conference of the Parties to the Convention on Biological Diversity, in decision X/28 paragraph 39, recognized the good synergies between the Convention on Biological Diversity and the Ramsar Convention on Wetlands and requested the Executive Secretary, and invited the Secretariat and Scientific and Technical Review Panel (STRP) of the Ramsar Convention, and other relevant partners, subject to the availability of financial resources, to establish an expert working group, building upon the relevant core expertise of the STRP, to review available information, and provide key policy relevant messages, on maintaining the ability of biodiversity to continue to support the water cycle. Progress with the work of the expert group was reported to the fifteenth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA). SBSTTA recommendation XV/5, section II subparagraph (b), requested the Executive Secretary of the Convention on Biological Diversity to make the report of the expert group available for the information of the eleventh meeting of the Conference of the Parties. Consequently, the Executive Secretary is hereby making the report of the expert group available.

2. This document is circulated in the form and languages in which it was received by the Secretariat of the Convention on Biological Diversity.

3. The summary report of the expert group is provided for the consideration of the eleventh meeting of the Conference of the Parties to the Convention on Biological Diversity as document UNEP/CBD/COP/11/30.

*UNEP/CBD/COP/11/1.

Report of the expert group on maintaining the ability of biodiversity to continue to support the water cycle

September 2012

Acknowledgements

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Contributors to the work of the expert group are acknowledged in Table 1.1 of this report.

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EXECUTIVE SUMMARY

For the current purpose, and that of the eleventh meeting of the Conference of the Parties to the Convention on Biological Diversity, the Executive Summary of this report is document UNEP/CBD/COP/11/30 (http://www.cbd.int/doc/?meeting=COP-11).

CHAPTER 1 Introduction

1.1 Background to the work

The tenth meeting of the Conference of the Parties to the Convention on Biological Diversity, in decision X/28 paragraph 39, recognized the good synergies between the Convention on Biological Diversity and the Ramsar Convention on Wetlands and requested the Executive Secretary, and invited the Secretariat and Scientific and Technical Review Panel (STRP) of the Ramsar Convention, and other relevant partners, subject to the availability of financial resources, to establish an expert working group, building upon the relevant core expertise of the STRP, to review available information, and provide key policy relevant messages, on maintaining the ability of biodiversity to continue to support the water cycle.

Subsequent to securing funding, STRP organised an inaugural meeting of the expert group in June 2011. Progress with the work of the expert group was reported to the fifteenth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA). SBSTTA recommendation XV/5, section II sub paragraph (b), requested the Executive Secretary of the CBD to make the report of the expert group available for the information of, and a summary report of its findings for the consideration of, the eleventh meeting of the Conference of the Parties. Consequently, the current document was prepared by the expert group to assist the Executive Secretary to fulfil this request. The aforementioned summary report of the expert group is provided to the eleventh meeting of the Conference of the Parties to the CBD as document UNEP/CBD/COP/11/30.

The work of the expert group was based on peer-reviewed scientific literature.

1.2 Ongoing work of the expert group

The expert group continues to further develop the current report as follows:

a) Provide a chapter on the hydrological functions of grasslands to complement existing chapters in order to adequately cover the major terrestrial biome types;
b) Derive the key gaps in the scientific knowledge (currently identified gaps are dispersed in this report and require consolidating across biomes);
c) Present the completed scientific findings to the Subsidiary Body on Scientific, Technical and Technological Advice, in particular regarding science gaps;
d) Publish the final report in the peer reviewed academic literature (most of the content of the current report has also been peer reviewed); and
e) Subject to resource availability, the STRP, Ramsar and CBD Secretariats will produce simplified versions of the content report for a more general readership.

1.3 Contributors to the work of the expert group

The contributors to the work of the expert group, as of the date of this report, are listed in Table 1.1.

Table 1.1: List of contributors to the work of the expert group, in alphabetical order.

Acreman, Mike. Centre for Ecology and Hydrology, Wallingford, United Kingdom
Albertengo, Juliana. No-Till Association of Argentina (Aapresid), Rosario, Argentina
Amado, Telmo. Federal University of Santa Maria, Santa Maria, Brazil
Amis, Mao. Pegasys, South Africa

Anderson, Aileen. Crossflow, South Africa

Bacchur, Isam. American University of Beirut, Lebanon

Basch, Gottlieb. University of Evora, Portugal

Binnie, Chris. United Kingdom

Calegari, Ademir. Agricultural Institute of Paraná (IAPAR), Brazil

Chappell, Nick A. Lancaster Environment Centre, Lancaster University, Lancaster, United Kingdom

Chettri, Nakul. ICIMOD, India

Coates, David. Secretariat of the CBD, Montreal, Quebec, Canada

Cohen-Shacham, Emmanuelle. Department of Zoology, Tel Aviv University, Tel Aviv, Israel

Corsi, Sandra. University of Teramo, Teramo, Italy

Davidson, Nick. Ramsar Secretariat, Gland, Switzerland.

de Mello, Carlos Rogério, Departamento de Engenharia, Universidade Federal de Lavras, Lavras/MG, Brazil

Fleiner, Renate. ICIMOD, Swirzerland

Friedrich, Theodor. FAO, Rome, Italy

George Lukacs, James Cook University, Townsville, Australia

Goddard, Tom. Alberta Agriculture and Rural Development, Edmonton, Alberta, Canada

Gonzalez-Sanchez, Emilio. University of Córdoba, Spain

Gregersen, Hans. Minnesota University, USA

Harwood, Richard. Michigan State University, USA

Hussein, Meral. University of Ottawa, Ottawa, Canada

Kassam, Amir. University of Reading, United Kingdom

Kim, Ik-Jae. Korea Environment Institute, Korea

Kim, Kwi-Gon. Emeritus Professor, Seoul National University, Seoul, Korea

Laurent, François. Université du Maine, France

Li, Hongwen. China Agriculture University, Beijing, China

McCartney, Matthew. International Water Management Institute, Vientiane, Lao P.D.R.

McInnes, Rob. Director, RM Wetlands & Environment Ltd, Oxfordshire, United Kingdom

Mello, Ivo. No-Till Federation of Brazil (FEBRAPDP), Ponta Grossa, Paraná, Brazil

Moreno-Casasola, Patricia. Red de Ecología Funcional, Instituto de Ecología, Veracruz, México

Nurbekov, Aziz. ICARDA, Tashkent, Uzbekistan

Okruszko, Tomasz. Division of Hydrology and Water Resources, Warsaw University of Life Sciences, Poland

Peiretti, Roberto. No-Till Association of Argentina (Aapresid), Rosario, Argentina

Pretty, Jules. University of Essex, Colchester, UK

Ralisch, Ricardo. University of Londrina, Brazil

Sá, Jõa Carlos Morales. University of Ponta Grossa, Brazil

Shahriar, M. Wahid. ICIMOD, Bangladesh

Sharif, Asif. Farm All Technology Pvt. Ltd, Lahore, Pakistan

Shrestha, Arun Bhakta. ICIMOD, Nepal

Sinun, Waidi. Conservation & Environmental Management Division, Yayasan Sabah Group, Kota Kinabalu, Sabah, Malaysia

Sturny, Wolfgang. Swiss No-Till, Ligerz, Switzerland

Thierfelder, Christian. CIMMYT, Zimbabwe

Uphoff, Norman. Cornell University, Ithaca, USA

Wani, Suhas. ICRISAT, India

Yakushina, Ekatarina. McGill University, Montreal, Quebec, Canada

CHAPTER 2 Forests

2.1 Scope of the assessment

When considering the *hydrological functions* associated with forests (Section 2.4) and the resultant impact on the delivery of *ecosystem services* (Section 2.6) it is often the activities that take place (or do not take place) within closed forests (or open woodland) rather than the impact of individual trees that require assessment. Thus the focus of this synthesis of hydrological functions and ecosystem services strictly should relate to ‘Forestlands’ (cf. ‘Wetlands’) rather than ‘Forests’, to capture both the effects of individual trees and the impacts of management practices on soils, water and microclimates within forested areas. The interaction of hydrological functions with forest functions for carbon capture and retention will be discussed separately in Section 2.6.

This assessment covers all global forests along the latitudinal gradient from boreal forest (50-60^{o} N) to temperate forest and then tropical forest (Foley *et al*., 2005). Tropical forests include small areas of Tropical Montane Cloud Forest.

2.2 Global extent of forests

Forest and woodland areas with more than 10 percent tree cover currently extend over 4 billion hectares or 31 percent of the land area of the globe (Fig. 2.1). FAO (2010) hase estimated that 65 percent of these forests are, however, in a disturbed state. Hansen *et al* (2008) suggests that this figure may be even higher for lowland evergreen rain forest in the tropics. Further disturbance is expected, given that some 30 percent of the world’s forests are classified as Production (rather than Protection) Forest where commercial forestry operations predominate; plus a further 16 percent of the world’s forests are unclassified (FAO, 2010) and likely to be subject to disturbances. Within some tropical regions, notably Asia, tree planting is off-setting the rate of forest loss. Within this region, newly forested areas now exceeded 120 million hectares (FAO, 2010). The global rate of reforestation and afforestation cannot, however, offset the net loss of 7-11 million km^2 (0.7-1.1 billion hectares) of closed forests over the last 300 years (Foley *et al*., 2005); this includes 2.4 million km^2 and 3.1 million km^2 lost from North America and Europe, respectively (Bryant *et al*., 1997). Indeed, Drigo (2004) calculated a ratio of 18-24: 1 for the balance between closed forest destruction to forest planting. Consequently, it is essential that that this synthesis properly quantifies the significance of findings pertinent to the globally extensive *disturbed natural forests* in addition to those studies from undisturbed natural forests and plantations.

Fig. 2.1 Extent of global forested area (> 10 percent tree cover) from FAO (2010).

2.3 Hydrological processes within forests

The subject of the interaction between forests and water is plagued by myths, misinterpretations and too hasty generalisations (Andréssian, 2004; Chappell, 2005; Tognetti *et al*., 2005). This problem is a century old, with Pinchot (1905) noting "…it is unfortunate that so many of the writing & talking upon this branch of forestry has had little definite fact or trustworthy observation behind it. The friend & the enemies of the forest have both said more than they could prove…" (cited in Andréassian, 2004). Part of the misconceptions and debate about the interaction between forests and water globally is due to the ambiguous or even incorrect use of hydrological terms. It is therefore essential that the section on the evidence for the hydrological functions of forests is preceded by a *precise scientific definition* of the *hydrological pathways* underpinning the hydrological functions of forests. *Unless the hydrological pathways are defined correctly, accurately quantified and not confused, then the hydrological functions of forests are likely to be grossly misinterpreted*. Hydrological pathways are also called 'water-paths', 'runoff pathways' and 'streamflow generation pathways' when referring to the pathways of water penetrating the forest canopy to travel on or beneath the forest floor towards a stream channel. Within this synthesis, the hydrological pathways within the forest canopy (i.e., rainfall and snowfall reaching the forest canopy, cloud water interception, wet-canopy evaporation, throughfall, stemflow and transpiration) are discussed in addition to the runoff pathways.

The hydrological pathways that may be present within a forest environment are shown diagrammatically within Fig. 2.2. The hydrological pathways shown are: A = rainfall and/or snowfall, B = horizontal (occult) precipitation capture, C = wet-canopy evaporation (or interception loss), D = transpiration, E = throughfall and stemflow, F = infiltration-excess overland flow, G = infiltration, H = lateral subsurface flow in soil strata, I = lateral subsurface flow in regolith and/or rock, J = saturation overland flow (including recharge by return flow), and K = riverflow (or channel flow).

Rainfall and/or snowfall (Path A): Rainfall is defined here as precipitation in liquid state received in a raingauge located at the top of the canopy forest canopy (or a canopy gap) and with a funnel facing vertically (as separated from an occult precipitation gauge). Within this study, use of the term 'rainfall' without any qualifiers only refers to 'gross rainfall', i.e., the rainfall received above any vegetation canopies, and not to 'net rainfall', which is the rainfall received beneath vegetation canopies (i.e., throughfall and stemflow combined). Snowfall is the depth of precipitation collected using a snow pillow or by the melting of snowfall into funnel facing vertically. Clearly this depth may be different to that preferentially trapped by a forest canopy and is particularly important in boreal forests (Suzuki and Nakai, 2008).

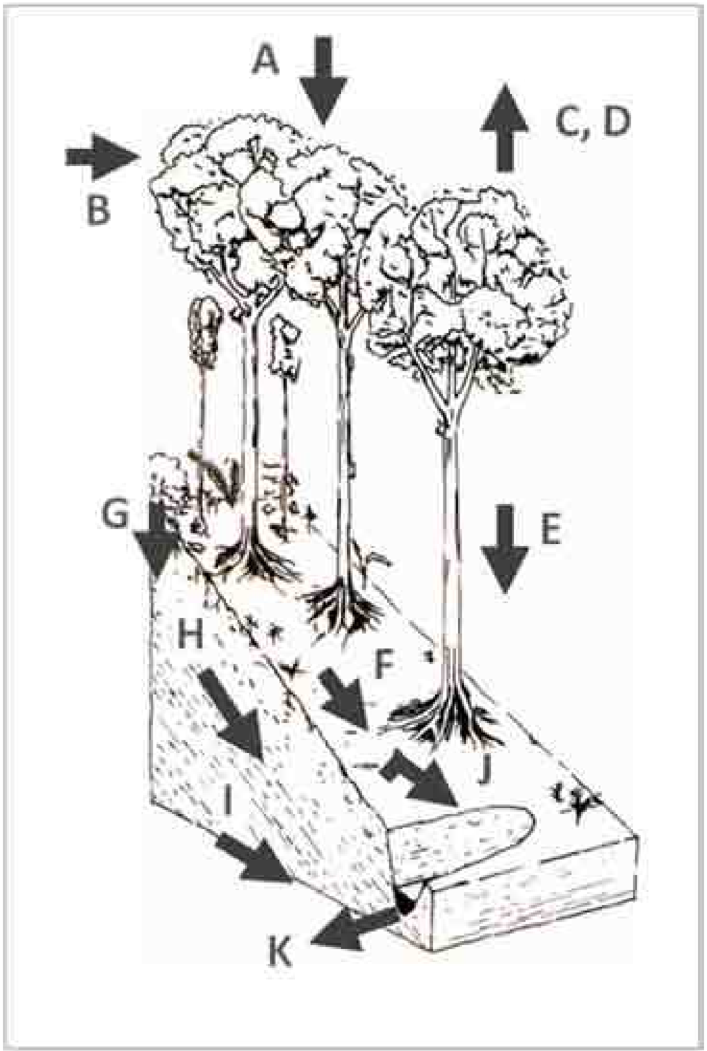


Fig. 2.2. Hydrological pathways are shown within a forested hillslope schematic, but present at scales from 0.1 km^2 experimental basins to international basins covering millions of square kilometres. Adapted by NA Chappell from the original diagram by Nick Scarle (with permission) published in Douglas (1977) Humid Landforms. MIT Press.

Horizontal (occult) precipitation capture (Path B): Horizontal or occult precipitation is that component of the precipitation measured using interceptor meshes that can capture occult precipitation (i.e., mist, fog etc: see Bruijnzeel *et al*., 2010).

Wet-canopy evaporation (Path C): Wet-canopy or wetted-canopy evaporation (Stewart, 1977) is the depth of water evaporated to the atmosphere from wetted parts of vegetation surfaces (i.e., leaves, branches and stems). This includes a forest canopy and a forest understory. Note that term wet-canopy evaporation is used in preference to ‘interception loss’ as ‘interception loss’ can be confused with ‘interception’, which means the water intercepted by a vegetation canopy, some of which will penetrate and reach the ground as throughfall and stemflow (see below), while some will leave the canopy as wet-canopy evaporation, and some stored temporarily on vegetation surfaces. Volumetrically, wet-canopy evaporation is most important in areas of low rainfall intensity, high rainfall totals, high wind run and forest canopies with a high leaf area index (Molchanov, 1960; Calder, 1990; Roberts *et al*., 2004).

Transpiration (Path D): Transpiration is the evaporation of water from within plant stomata into the atmosphere. This process is supported by water abstracted from the soil by plant roots and transported to the stomata within plant xylem.

Throughfall and stemflow (Path E): Throughfall is the component of the 'gross rainfall' (sometimes with some occult precipitation) that penetrates a vegetation canopy by either falling through gaps between branches and leaves or by hitting a branch or leaf before then falling to the ground. Stemflow is the component of the 'gross rainfall' that reaches a branch and then travels along to a plant stem on its way down to the ground surface. After integration of several days of data, the gross rainfall minus the net rainfall is equal to the wet-canopy evaporation.

Infiltration-excess overland flow (Path F): When the rainfall intensity (e.g., mm/15-mins or mm/hr) exceeds the saturated hydraulic conductivity of the ground surface (equal exactly to the infiltration capacity, and also in mm/15-mins or mm/hr) then infiltration-excess overland flow will occur either on (Horton, 1933) or laterally within the forest litter layer (Hewlett, 1982). This hydrological pathway has been considered by engineers (civil, agricultural and hydrological) for the last 80 years to be the dominant pathway of water to rivers during rainstorms. Experimental evidence collected over the same period (by experimental hydrologists, forest hydrologists, hillslope hydrologists and scientific hydrologists) does, however, show that this pathway is a <u>volumetrically insignificant component of the river hydrograph</u> (Hursh and Brater 1941; Dubreuil, 1985; Chappell *et al*., 2006), except for a few isolated locations. Simply, the saturated hydraulic conductivity of the ground surface beneath most vegetated surfaces (forest, grass or crops) is far in excess of the <u>dominant</u> rainfall intensity at most locations. The exceptions occur in areas of slowly permeable soils (e.g., FAO Gleysols, FAO Vertisols), particularly where they coincide with areas of very high rainfall intensity (e.g., areas beneath the tracks of tropical cyclones). Intense compaction of topsoil by vehicles (Ziegler *et al*., 2007) or livestock trampling (Bonell *et al*. 2010) can also locally reduce the infiltration capacity sufficiently to give locally significant volumes of overland flow. While the infiltration-excess overland flow pathway may not transport most of the water that reaches the most rivers, it is of fundamental importance to the transport of soil particles (and bound chemicals such as phosphorus or pesticides) during the process of soil erosion (see section 2.6 and 2.8).

Infiltration (Path G): The movement of water into the topsoil (or ground surface where soil development is absent) is defined as the infiltration (cf. Hewlett, 1982 definition of infiltration-excess overland flow). In most areas of the globe at most times, rainfall (gross or net) is able to infiltrate the topsoil.

Lateral subsurface flow in soil strata (Path H): Once water has entered the soil by infiltration, it may then percolate vertically into underlying strata of unconsolidated rock (e.g., saprolite, glacial till) or a solid rock aquifer (i.e., a rock with both a high saturated hydraulic conductivity and porosity), where either are present. Alternatively, all or a proportion of the percolation may be lateral (i.e., downslope) within the A and B soil horizons to emerge in a river channel (or prior to a channel via 'return flow': Cook, 1946).

Lateral subsurface flow in unconsolidated rock and/or solid rock (Path I): Where deep strata of unconsolidated rock are present (e.g., granite saprolite), and are between a permeable A and B soil horizon and a impermeable rock strata, then lateral flow towards a river can take place with this layer. If the solid rock has a high saturated hydraulic conductivity and porosity (a rock aquifer by definition) and lies beneath permeable overlying horizons, then the dominant lateral flow towards the river will be within the rock. These deeper hydrological pathways tend to have a slower response to rainfall in comparison to the shallower pathways in the A and B soil horizons. Lateral flows within unconsolidated rock and/or solid rock aquifers can be described as 'groundwater', though care is needed, as hydrogeologists use this term to describe only flow within the permanently saturated strata. The role of these deeper pathways in streamflow generation (Hursh and Brater, 1941), have been incorrectly ignored by many studies (Bonell, 2004).

Saturation overland flow (including recharge by return flow) (Path J): Where subsurface flow (within Path H and/or I) emerges from the ground prior to reaching a channel ('return flow') then it will flow over the surface as saturation overland flow. In these 'wetland' areas, overland flow may be present where the prevailing rainfall intensity is less than the local saturated hydraulic conductivity. Any rainfall falling onto these saturated topsoils with their upward return flow will not be able infiltrate, and so add to the volume of saturation overland flow travelling towards the nearest river channel. Because subsurface flows tend to converge on channels, the riverside (or 'riparian' or valley bottom) soils have a greater likelihood of generating saturation overland flow (Kirkby, 1976).

Riverflow (or channel flow) (Path K): Once water from overland and subsurface pathways (Paths F, H, I and J) enters a defined river channel it then becomes riverflow. This hydrological pathway is responsible for the transport of water, particles and solutes over long distances within landscapes whether covered by forest or other land-uses. Strictly, the term *runoff* is the river discharge per unit basin area (e.g., units of mm/hr), particularly within water budget and modelling studies. Use of this term is, however, avoided because of the ambiguity arising from its alternative use to described rapid overland (Paths F and J) and rapid subsurface pathways (Path H and sometimes Path I also).

2.4 Observed evidence for the hydrological functions of forests

Any review of the observed evidence for the hydrological functions of forests has to manage the huge wealth of literature on certain topics, in addition to managing the problem of myths and misinterpretations noted earlier. Some topics, notably the effects of forest on the available water resources in rivers ('annual water yield') have received much study, while the effects of forested areas (forestlands) on water quality (relative to that of other land uses) have received comparatively little study (Chappell *et al*., 2007). Given these issues, several guiding principles have been established to structure the review and synthesis of the findings from boreal, temperate and tropical environments.

2.4.1 Guiding principles for reviewing the observed evidence of the hydrological functions of forests

The synthesis attempts to identify all hydrology-mediated processes operating in natural forests (of boreal, temperate and tropical environments) and plantations. Given the political significance of carbon capture and retention globally, and its link to the hydrological functions of forests, this additional forest function will be incorporated within the overarching perceptual model or schematic diagram (Fig. 2.4) and in a separate discussion (Section 2.5).

All of the hydrological functions to be identified *must be capable of being linked unambiguously to specific hydrological pathways* (Section 2.3), and consideration of the relative importance of each hydrological function globally must be consistent with the relative importance of each hydrological pathway involved.

This *synthesis is explicitly based on rigorous observational evidence* (i.e., well-designed field studies where observed data and findings have been subject to peer review). Ideas, views, concepts and models are only discussed where they relate to the observed evidence, or its absence.

Much observed evidence of the hydrological function of forests has come from direct comparison of the behaviour of forested basins (in response to tree cutting or planting) with that of adjacent non-forest basins. These so called *paired-catchment studies* (or paired-watershed studies in the USA) have been and remain important to the study of forest-water interactions (Swank and Crossley, 1988; Webb *et al*., 2012).

Some reviews of the hydrological function of forests focused only a limited number of beneficial effects to ecosystem services, while others have focused on a similarly limited number of negative effects to ecosystem services (e.g., Hayward, 2005). It is important that a *balanced approach* presenting the *dominant mode of behaviour of all hydrological functions of forest*, whether resulting in positive or negative effects to ecosystem services, is given (Bruijnzeel, 1990; Chappell, 2006; Chappell *et al*., 2007). Notable exceptions (often location specific) to the dominant mode of behaviour should be presented, particularly given that the decision to designate a dominant mode of behaviour is partly subjective. Additionally, these exceptions (or atypical responses of a hydrological function) need to be known where new management options (Section 2.8) or policies (Section 2.9) are to be advocated.

To facilitate a balanced view of the observed evidence for the range of hydrological functions within forests, a 'traffic light schematic' is used to summarise the findings for each hydrological function. The schematic diagram (Fig. 2.3) shows: (1) whether the forest impact on a specific hydrological function is broadly positive (green circle) or negative (red circle) for the dominant ecosystem service delivered, (2) the strength of evidence of the observed impact globally (i.e., from one circle for very few rigorous studies to three circles for numerous rigorous studies in different global forests), and (3) the global extent of the impact. Where the forest impact on a function is specific to a small area of globe (e.g., cloud forests), then this is shown with small circles. Where the studies indicate that the impact is widespread across the globe, then this is shown with large circles (Fig. 2.3).

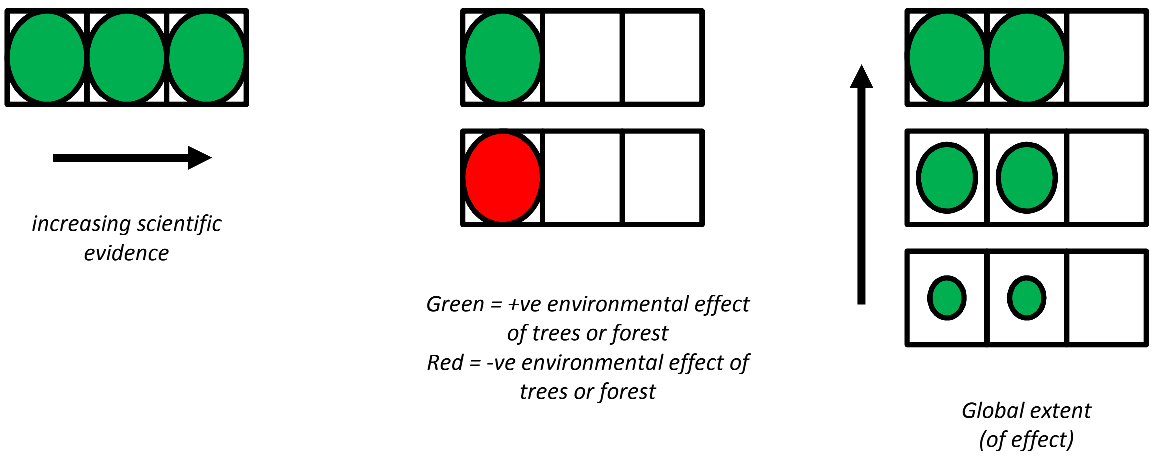


Fig. 2.3 A traffic light system for summarising the observed evidence for the effect of forests on hydrological functions

2.4.2 Summary schematic of the hydrological functions of forests

The schematic diagram illustrating the possible hydrological functions of forests (but not their magnitude, level of evidence or spatial extent) is given in Fig.2 .4. These functions are shown in a way that illustrates their linkage to the underlying hydrological pathways (Fig. 2.2). The direct hydrological functions shown are the: water availability via evaporation function (F1), precipitation capture function (F3), microclimate function (F4), enhanced infiltration (and reduced overland flow on slopes) function (F5), reduced slope erosion function (F6), exclusion of pollutant inputs function (F7), downslope utilisation of leached nutrients function (F8), downslope (and coastal) physical function (F9), peak-flow damping and low-flow enhancement function (F10), and enhancement of river water quality function (F11).

The carbon dioxide capture function (F2), and aquatic carbon source function (F12) is also shown to allow this schematic to illustrate the links to the hydrological pathways and functions, but is discussed separately in Section 2.5.

The observed evidence for each of the hydrological functions of forests will be discussed in the following sub-sections.

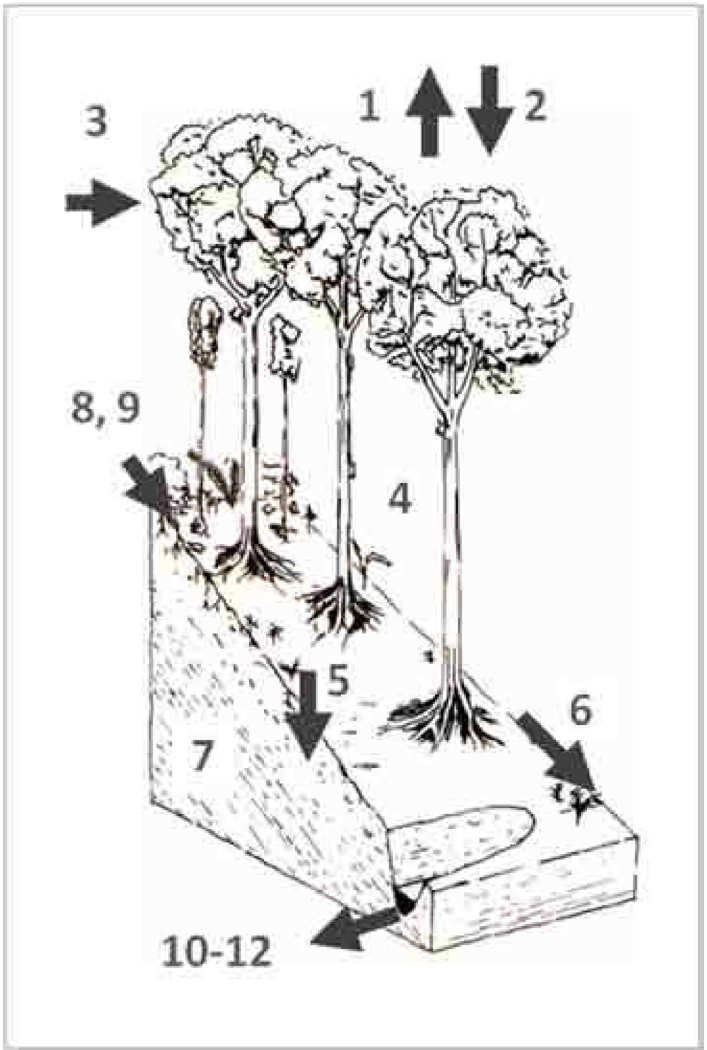


Fig. 2.4. Hydrological functions of forested areas, and include the important fluxes of gaseous, dissolved and particulate carbon that are transported by latent heat flux and riverflow. Each function is numbered where 1 = Water availability via evaporation function (-/+ve), 2 = Carbon dioxide capture function (+ve), 3 = Precipitation capture function (+ve), 4 = Microclimate function (+ve), 5 = Enhanced infiltration (& reduced overland flow on slopes) function (+ve), 6 = Reduced slope erosion function (+ve), 7 = Exclusion of pollutant inputs function (+ve), 8 = Downslope utilisation of leached nutrients function (+ve), 9 = Downslope (and coastal) physical function (+ve) 10 = Peak-flow damping and low-flow enhancement function (+ve), 11 = Enhancement of river water quality function (+ve), 12 = aquatic carbon source function (+ve).

2.4.3 Water availability via evaporation function

The presence of trees (rather than herbaceous vegetation, crops or bare ground) may affect the annual availability of water resources within deep groundwater (Path I) or rivers (Path K). The effect of trees on the provisioning ecosystem service of water supply (i.e., the water people abstract from the environment) is primarily via the evaporation function. Trees and forests affect the evaporation function via changes to the wet-canopy evaporation pathway (Path C) and/or transpiration pathway (Path D). The total evaporation is known by the American term ‘evapotranspiration’.

A huge number of paired-catchment studies have been used to examine the effects of removing trees from boreal, temperate and tropical forests and the effects of planting trees on the annual water yield of rivers. These studies have addressed forest clearance ('deforestation') and the selective logging practices characteristic of many tropical forests. Many studies show that natural forests and plantations have higher rates of evaporation than nearby herbaceous vegetation and so leave less water resources available in rivers. The higher evaporation relates to: (1) deeper tap roots that are able to continue to abstract water during dry seasons (Canadell *et al*., 1996), (2) a greater leaf area index, particularly with conifers, giving greater rates of wet-canopy evaporation (Calder, 1990), and (3) high growth rates and lower water use efficiency for young plantation trees (Vertessy *et al*., 2003). Rigorous reviews (e.g., Andréssian, 2004) of the available studies have however shown that the impact of the removal or addition of trees from the same catchment proportion gives very different changes in the annual river discharge per unit area (mm; Fig. 2.5). Some studies show large reductions of water yield in rivers, while others show only small or no changes.

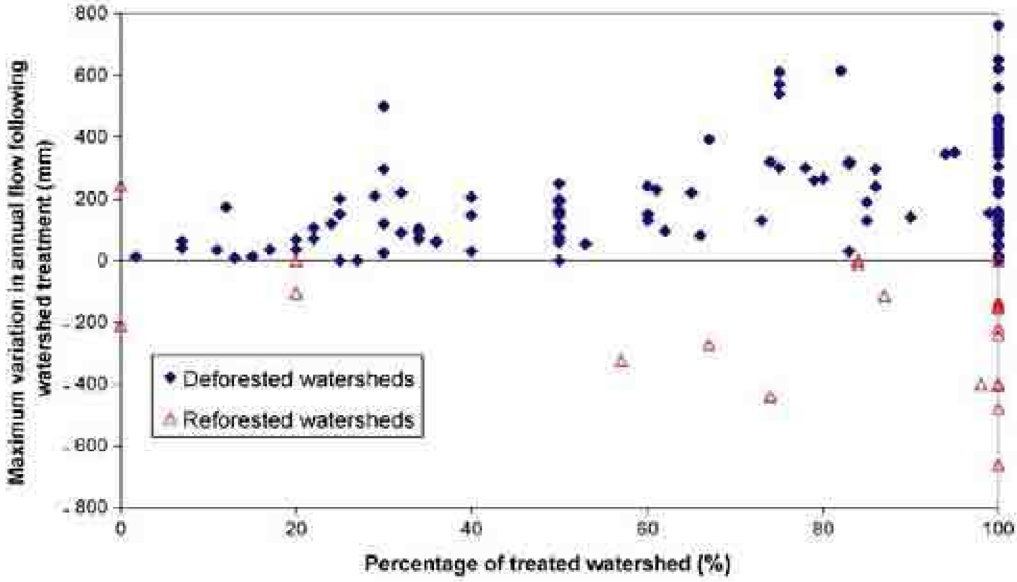


Fig. 2.5 Changes in the annual river discharge per unit area due to tree removal or planting, from Andréssian (2004 J. Hydrol. 291: 1-27)

There are indications that differences in 'tree type' affects evaporation. The review of Brown *et al*. (2005) that focused on forests in boreal and temperate locations, showed that conifers generally used (transpired) more water than hardwood trees (Fig. 2.6). They also noted that assessments were very sensitive to whether the 'peak change in water yield' or an 'average change over the duration of the study' was used.

Perhaps the clearest findings of the impact of trees on the evaporation function are from the study of Zhang *et al*. (2001). They reviewed 250 catchment-based, water balance studies from across the globe, including 35 from countries within the humid tropics. They demonstrated that the difference (mm) between water use by forests relative to that by grassland increases as the annual rainfall (mm) increases. Their model, fitted to the large number of data-sets with a correlation coefficient of 0.96, showed that forests typically have much greater evaporation rates than grasslands where rainfall exceeds 2000 mm/yr, but comparable rates where rainfall is less than 500 mm/yr (Fig. 2.7).

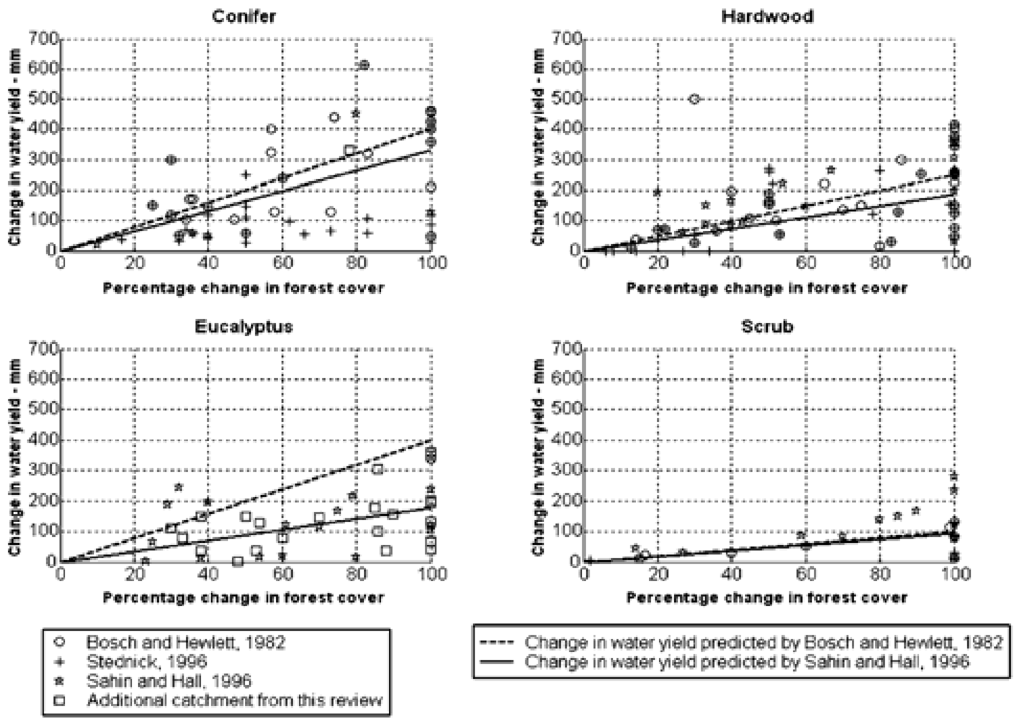


Fig. 2.6 Changes in the annual river discharge per unit area due to tree removal or planting from Brown *et al*. (2005 J. Hydrol. 310: 28-61)

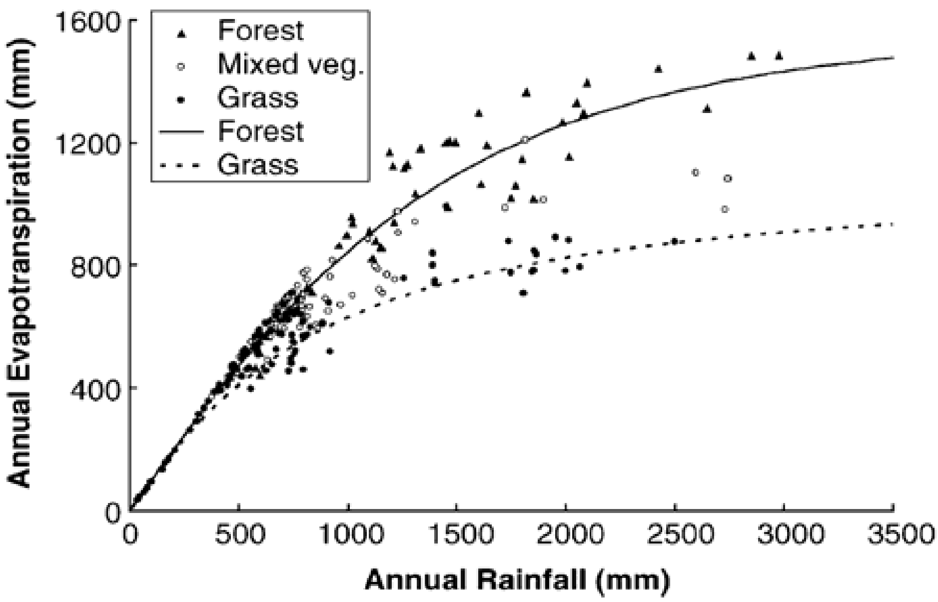


Fig. 2.7 Evaporation by forest and grassland basins (mm/yr) against annual rainfall from Zhang *et al* (2001 Water Resources Res 37: 701-708)

In contrast to the large number of studies that have examined forest impacts on annual river yield, very few observational studies have examined the impact on deep groundwater resources (Zhang and Schilling, 2006). There is also a lack of rigorous field studies that compare forest water use against that by irrigated crops and theoretically, irrigation in once forested areas may offset the effects of forest removal (see Ozdogen *et al*. 2010). This is particularly important in the seasonal tropics regions where rates of potential evaporation are very high.

As forest cutting may locally reduce the amount of water returned to the atmosphere, rainfall totals produced by local re-cycling may be affected. Extensive observational evidence for this effect is however lacking (Bruijnzeel, 2006). In part this is because a significant proportion if the rainfall over land is derived from ocean evaporation (Goessling and Reich, 2011) and partly because of the difficulty in attributing decadal changes in rainfall to land-cover change rather than the effects of natural climate dynamics (Chappell and Tych, 2012). The study of Lawton *et al*. (2001) does however, present observations to show that deforestation of lowland rainforest in Costa Rica has reduced local cloudiness. They then show by simulation, that this may affect rainfall (Path A) and horizontal precipitation (Path B) in downwind cloud forest. Many more observation-based studies such as Lawton *et al*. (2001) are however needed to show the true role of deforestation on local moisture recycling.

Given that a reduction in surface-water or groundwater resources for water supply abstractions is a negative impact on this provisioning ecosystem service (and is particularly important in the dry season: Avila, 2011), then most observational evidence indicates that the impact should be considered as negative for high rainfall regions (Fig. 2.8). Given the number of studies collated by the reviews, this observed evidence is considered to be well attested for such regions. There is also no reason to believe that this phenomenon does not apply across large areas of the globe (i.e., humid tropical or humid temperate environments). The studies conducted in relatively dry regions of the globe and reviewed by Zhang *et al*. (2001), do however show no or little difference in water use by forest versus grassland. This finding indicates a neutral impact of forests on water resources ('an orange traffic light': Fig. 2.8). By incorporating the potential benefits of forests to local moisture recycling may indicate that the overall impact of forests on water availability is closer to neutral, than many reviews would suggest.

A further exception to the impacts summarised in Fig. 2.8 is the localised impact of forests within riparian zones (Sections 2.2.4.9 and 2.2.4.10) of high rainfall areas. Here the greater evaporation resulting from the presence of forest might be seen as a positive impact, as greater soil drying can reduce the amount of saturation overland flow (Path J) and hence reduce the transport of chemicals (Sections 2.2.4.9) and soil particles (Sections 2.4.10) to rivers (Hernandez-Santana *et al*. 2011).

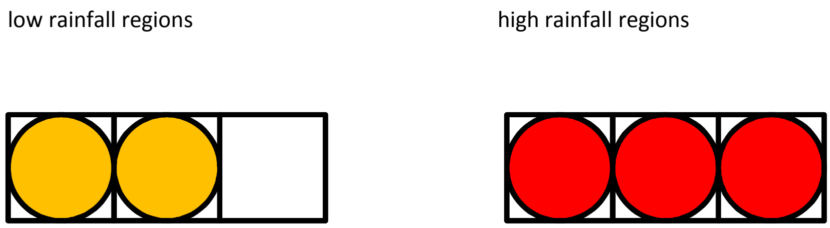


Fig. 2.8 Summary of the findings on *water availability via evaporation function* of forest (see Fig. 2.3 for key)

2.4.4 Precipitation capture function

Very high altitude areas are frequently in cloud. Forests within these areas are more efficient at capturing cloud water than low herbaceous vegetation. This means that forests are better at capturing water from within clouds. This process is now called *Cloud Water Interception* (CWI: Bruijnzeel *et al*. 2010), and formerly 'fog interception'. During CWI measurement, *Horizontal (wind-driven) Rainfall* is also captured (both give Path B of Section of 2.2.3). Locally, rates of CWI can be very high, for example Juvik and Nullet (1995) recorded throughfall beneath Tropical Montane Cloud Forest that was 120-180 percent of the open site (vertical) rainfall. The areas of Tropical Montane

Cloud Forest that are able to capture significant volumes of rainfall by this mechanism only occupy 215000 km^2 of the globe or 1.4 percent of tropical forest (Bruijnzeel *et al*. 2010).

Some boreal forests are in areas with a significant proportion of precipitation received as snowfall. In comparison to herbaceous vegetation, the higher canopies and leaf area index of forests can capture more horizontal and drifting snow.

Where the enhanced capture of precipitation by forested areas can be better utilised than if it fell elsewhere (e.g., at sea: Prada *et al*., 2010), then this function should be considered as a positive impact on the provisioning ecosystem service of water supply. The area of the globe where forests can enhance the capture of cloud water is however small (Fig. 2.9).

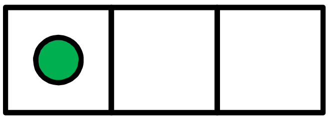


Fig. 2.9 Summary of the findings on the *precipitation capture function* of forest (see Fig. 2. 3 for key)

2.4.5 Microclimate function

Closed forests and open woodlands provide shade from direct solar radiation, shelter from rainfall and wind, plus regulate the humidity and temperature (soil and air) beneath their canopy (Gardiner *et al*., 2006; UK National Ecosystem Assessment, 2011). Most observational evidence comes from ecological studies along forest edges (e.g., Pinto *et al*., 2010; de Siqueira *et al*., 2004; Davies-Colley *et al*. 2000).

Tree shelterbelts (and forest edges) also affect the microclimate of adjacent land and can positively affect livestock production and the growth of crops through reduced evaporation (Delwaulle, 1977; Brenner, 1996) and increased soil moisture (Muthana *et al*. 1984; Ujah & Adeye, 1984). These changes positively affect the provisioning ecosystem service of food production.
The impact of trees on microclimate positively affect human comfort in villages and towns (Handley and Gill, 2009) and so the regulating (and associated cultural/recreational) ecosystem service of the mitigation of climate stress. Additionally, riparian trees regulate stream temperature (Studinski *et al*., 2012) and so enhance the supporting ecosystem service of aquatic habitat (see Section 2.4.10).
The findings from the observed evidence for the microclimate function of forest is positive and should be extensive, but have not been collated systematically (Fig. 2.10).

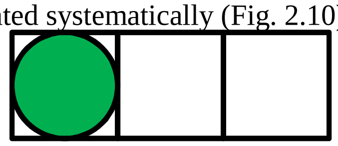


Fig. 2.10 Summary of the findings on the *microclimate function* of forest (see Fig. 2.3 for key)

2.4.6 Enhanced infiltration (& reduced overland flow on slopes) function

There is clear observational evidence that infiltration capacity of forest topsoil (equivalent to the saturated hydraulic conductivity of the topsoil: see Section 2.2.3) is typically greater than that of adjacent topsoil beneath grassland. This difference may be partly explained by the presence of a deeper litter layer, greater organic matter inputs to the topsoil and an absence of livestock trampling within most forests. Chandler and Chappell (2008) provide a table showing that the ratio of these two values is always larger than 1, and often considerably larger (Fig.2.11). More recently, Alvarenga *et al*. (2011) report a ratio of 5-15 for Cambisol soil beneath *Miconia sellowiana* trees relative to that beneath grassland.

With a higher infiltration capacity, there is an even greater likelihood that almost all net rainfall will infiltrate beneath forests, and minimise the production of infiltration-excess overland flow (Path F in Section 2.2.3). The greater infiltration will reduce the rate of soil erosion on slopes (having a mitigating impact on slope erosion: Section 2.4.7) and thereby enhance the regulating ecosystem services of reduced soil loss and enhanced water quality of rivers. These services have indirect impacts on provisioning services of food production and water supply, respectively.
If the presence of trees in a landscape with deep groundwater pathways (Path I; Section 2.3) can markedly reduce the proportion of the riverflow (Path K) that is generated by infiltration-excess overland flow (Path F), then the enhanced infiltration could add greater water resources to deeper groundwater reserves used for water supply or the slower hydrological pathways that maintain seasonal rivers during low-rainfall seasons (see the low-flow enhancement function in Section 2.4.10). This function would enhance the provisioning service of water supply. Few rigorous studies have addressed the water resource significance of the infiltration function, and new studies are needed, particularly within seasonally dry areas (Bruijnzeel 2004).

F/G	Soil type[a]	Tree type[b]	Reference
2.0	Luvisol	*Eucalyptus* spp.	Lorimer and Douglas (1995)
2.5	nk	*Eucalyptus* spp.	Burch et al. (1987)
3.4[c]	Gleysol	*Quercus robur*	This study
4.8	nk	*Pinus insularis*	Costalles (1979)
5.2	nk	*Pinus halepensis*	Berglund et al. (1981)
4.5–7.2	Cambisol	*Quercus robur*	Burt et al. (1983)
2.3–12	Ferralsol	*Eucalyptus/Gravillea* spp.	Wood (1977)
14	Nitisol	*Hibiscus elatus*	Ternan et al. (1987)
20	Andosol	Podocarp	Jackson (1973)
23–41	nk[d]	*Quercus* spp.	Molchanov (1960)
50[e]	Ultisol	*Quercus* spp.	Hoover (1949)
17–140	Cambisol	*Eucalyptus* spp.	Wood (1977)

F/G = ratio of the topsoil saturated hydraulic conductivity under trees to that under pasture (ranked by magnitude). (nk) not known.
[a] FAO-UNESCO classification.
[b] Dominant or representative tree species.
[c] At 3 m from Tree No. 1.
[d] Reported as 'dark grey soils'.
[e] 0.1 m depth.

Fig. 2.11 Ratio of the saturated hydraulic conductivity of the topsoil (A soil horizon) under forest compared to that under grassland for 12 studies reviewed by Chandler and Chappell (2008 For. Ecol. Manage. 256: 1222-1229; see this paper for the references cited therein).

Where soil infiltration capacity is very high beneath a pasture or cropland, so that virtually no infiltration-excess overland flow is produced, then the addition of trees may have a measureable impact on the infiltration capacity, but no measureable impact on the infiltration-excess overland flow (Gilmour *et al*., 1987). Equally, it should be noted that the beneficial effects of forests on infiltration are not as great, where there are is a high livestock density and hence marked soil trampling and compaction within forests (Bonell *et al*., 2010).
Despite these exceptions, there is ample observational evidence that the infiltration function of forests is clear, positive and extensive (Fig.2. 12).

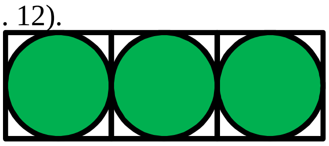


Fig. 2.12 Summary of the findings on the *infiltration function* of forest (see Fig.2.3 for key)

2.4.7 Reduced slope erosion function

There are numerous studies showing that undisturbed natural forest has a lower rate of slope erosion in comparison to cropland disturbed (tilled) on a regular basis. A recent study that demonstrates this reduced slope erosion function of forests is Liu *et al*. (2005) who used plot-scale measurements in Sichuan, China.

As a consequence, these forests maintain river water quality, notably lower turbidity and lower levels of pesticides and faecal coliforms that are transported with the soil particles (see Sections 2.4.8, 2.2.4.9 and 2.2.4.11). Elevated erosion also leads to accelerated losses of particulate carbon from slopes to rivers (Sections 2.5.2).

Studies have also reported localised reductions in slope erosion as a direct result of tree planting and growth via the beneficial impact on infiltration and soil stabilisation. One such study is that of Vasquez-Menandez *et al*. (2010) conducted in semi-arid Mexico.

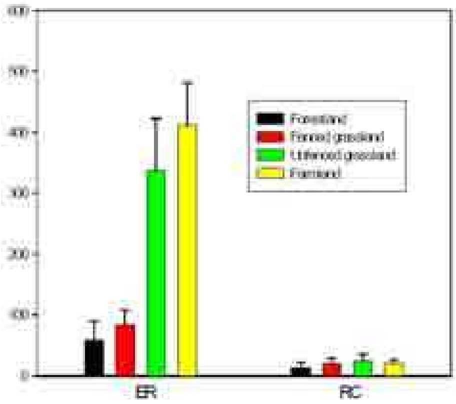


Fig. 2.13 Example of differences in slope erosion (ER) between undisturbed forest and cropland (labelled as 'farmland') by Liu *et al*. (2005 J. Mt. Sci-Engl: 2: 68-75). The runoff coefficient (RC) is also shown.

Where plantation development is accompanied by soil disturbances associated with artificial drainage, then the effects of forests in soil erosion may be negative. For example, artificial drainage prior to the planting of conifers in temperate Wales accelerated the rate of erosion within the studied Hafren and Tanllwyth basins in comparison to the pasture control – the Cyff basin (Fig. 2.14).

Catchment	Area (km^2)	Bed-load yield ($t\ km^{-2}\ y^{-1}$)	Suspended load ($t\ km^{-2}\ y^{-1}$)	Land use
Hore	3.08	11.8	24.4	Mature forest – first year of felling operations
Hafren	3.67	NA	35.3	Mature forest
Tanllwyth	0.89	38.4	12.1	Mature forest
Cyff	3.13	6.4	6.1	Pasture

NA = not available

Fig. 2.14 Bedload and suspended load resulting from erosion at the Plynlimon catchments, upland Wales (from Kirby *et al*. 1991 IoH Report 109).

Logging of forests (including the associated road construction in previously undisturbed natural forests) significantly accelerates erosion. Even selective harvesting of tropical natural forests gives increases in suspended sediment load that are between 4.3 and 52 fold larger than adjacent undisturbed forest basins (Chappell *et al*., 2004). During these harvesting periods the rates of erosion may be larger than those from nearby pastureland, though there is a dearth of such comparative studies.

The key message is that forests not subject to timber harvesting operations or artificial drainage have lower erosion rates than land covers subject to regular disturbance (e.g., tillage or livestock trampling), that has a beneficial impact on the regulating ecosystem services of reduced soil loss and enhanced water quality of rivers (Fig. 2.15). These services have indirect impacts on provisioning services of food production and water supply, respectively (as noted in Section 2.4.6). This beneficial hydrological function relates to the effect of: (1) greater soil surface protection, (2) enhanced infiltration (Path G) and (3) reduced infiltration-excess and saturation overland flow (Paths F and J), all via greater root development and litter-fall. If, however, the forest is subject to soil disturbances due to artificial drainage or harvesting, the presence of forests may have a negative impact on the soil erosion function (Fig. 2.15). The key issue may be the presence or absence of soil disturbance rather than trees.

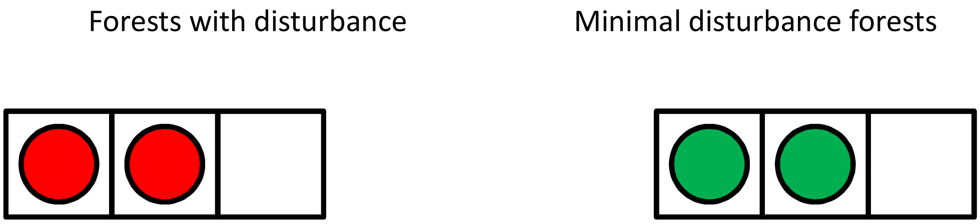


Fig. 2.15 Summary of the findings on the *reduced slope erosion function* of forest (see Fig. 2. 3 for key)

2.4.8 Exclusion of pollutant inputs function

Large global areas of cropland, grassland or urban development have large inputs of artificial chemicals (e.g., nitrates, phosphorus, pesticides) and/or artificial biochemicals (e.g., faecal coliforms associated with livestock or population centres), while these inputs tend to be absent from most areas of natural forests, and many plantations (Evans, 1982 p377; Chappell , 2005). This function clearly relates to the management practices within forest lands, rather than to the effects of individual trees (see Section 2.1). Liu *et al*. (2010) have shown that 136.60 trillion grams of nitrogen is added to the world's cropland each year; almost half as mineral nitrogen fertilizers. They also demonstrate that two fifths of this nitrogen is 'lost in ecosystems' – see Fig. 2.16 (i.e., stored or transported along overland, subsurface or river pathways: Paths F, H, I, J and K in Fig. 2. 2). Similarly, Macdonald *et al*. (2011) demonstrate that the agronomic input of phosphorus (P) in fertilizer amounts to 14.2 Tg P / yr globally, and a further 9.6 Tg P / yr is added as manure. They show that only 12.3 Tg P / yr are removed in crops, leaving a major imbalance and hence storage or transport along overland, subsurface or river pathways: Paths F, H, I, J and K in Fig. 2.2. Microbial contamination of water resources (e.g., faecal coliforms or cryptosporidium) is also an issue within non-forest areas. For example, Bolstad and Swank (1997) demonstrated how low levels of faecal streptococcus within the Coweeta forested catchment (USA) increased downstream, as urban development increased.
The absence or exclusion of large inputs of artificial chemicals or microbial contaminations within most natural forests means that groundwater (Path I) and river-water (Path K) draining from these forests *dilutes* the effects of contaminated drainage from the other land-uses. This exclusion of pollutant inputs function of forests consequently has a large positive impact on the provisioning ecosystem service of the supply of clean water suitable for abstraction and subsequent treatment for drinking water. It also has a positive impact on the regulating services of purifying soils and waters

and hence reducing human risk from contaminated waters, and the supporting service of providing river-water capable of sustaining life and biodiversity.

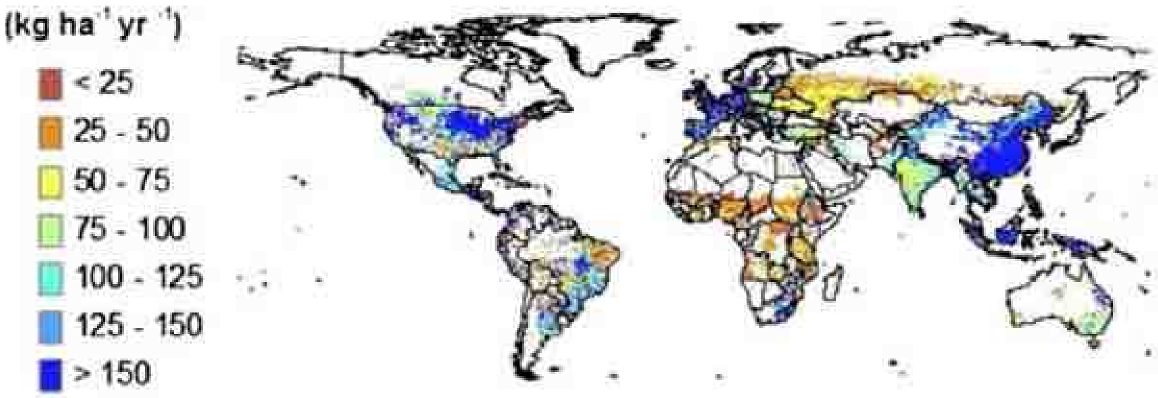


Fig. 2.16 Nitrogen outputs from cropland globally (Liu *et al*. 2010 PNAS 107: 8035-8040).

There are exceptions. Tree-crop plantations, e.g., oil palm, have high inputs of fertiliser and pesticides (Halimah *et al*., 2010). Similarly some commercial forests in the USA and elsewhere are treated with pesticides. Some agro-forestry systems in the tropics e.g., India, have high livestock densities and hence risks to water resources from microbial contamination. Indeed, many of the conifer plantations within the catchments of water supply reservoirs in the UK were added to exclude the risk of microbial contamination that was perceived to exist with the former land-use of grassland supporting cattle.

Given that the exclusion of pollutant inputs function applies to most natural forests (and these dominate globally: Section 2.2), its positive impact should be considered extensive globally (Fig. 2.17). The lack of rigorous studies that illustrate the effects on water quality of low-input forests versus high-input land-uses does however need to be highlighted. The observation that some forests, notably tree-crop plantations can have high chemical inputs also needs to be highlighted, even though they may be much less extensive (Fig. 2.17).

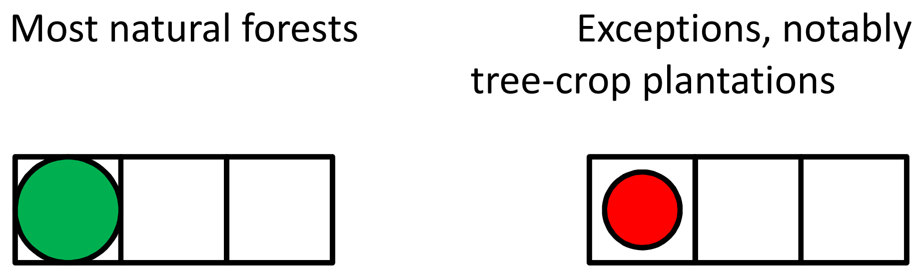


Fig. 2.17 Summary of the findings on the *exclusion of pollutant inputs function* of forest (see Fig. 2.3 for key)

2.4.9 Downslope utilisation of leached nutrients function

There has been considerable research into the value of trees in riverside or flood-plain areas (often called 'riparian areas') in mitigating water quality issues resulting from other land-uses upslope. Given the appropriate moisture regime and supply of carbon, trees within downslope areas can utilise nutrients leaching from upslope areas via overland or subsurface pathways (Paths F, H, I and J: Fig. 2. 3). High inputs to downslope areas often result because of the high artificial fertiliser inputs to cropland or improved pasture in the upslope areas. The widely cited study of Peterjohn and Correll (1984) undertaken in Maryland (USA) clearly demonstrates how downslope forest can remove dissolved nitrogen from overland flow (Paths F and/or J: that they called 'runoff': Fig. 2.18) and subsurface water (Paths H and/or I: that they called 'groundwater': Fig. 2. 18) draining from cropland.

Riparian forest has also been used successfully to reduce nitrate levels in contaminated rivers by diverting some of the riverflow into riparian forest via irrigation channels, to then return via drainage channels (Gumiero *et al*. 2011). Additionally, the negative effects on river nitrate loads of forestry drainage and logging operations with commercial forests have been reduced by drain blocking within riparian forest (Hynninen *et al*., 2011) or preventing harvesting of riparian forest (Clinton, 2011).
Given the importance of carbon to denitrification and to food webs, the enhanced litter-fall under riparian forest compared to other vegetation covers has been shown to be a beneficial function (Newham *et al*. 2011).
Critically, the effectiveness of riparian forest in the function of chemical removal from surface and subsurface waters is site specific, being dependent on: (1) the local biogeochemical conditions e.g., carbon availability and (2) the local hydrological conditions e.g., soil moisture content and hydrological pathways (Burt *et al*., 2010). Consequently, this hydrological function can have neutral or positive impacts on the same provisioning, regulating and supporting ecosystem services as the exclusion of pollutant inputs function (Section 2.4.8). Systematic global analysis of the extent of the conditions conducive to the effectiveness of this riparian forest function is however needed (Fig. 2.19).

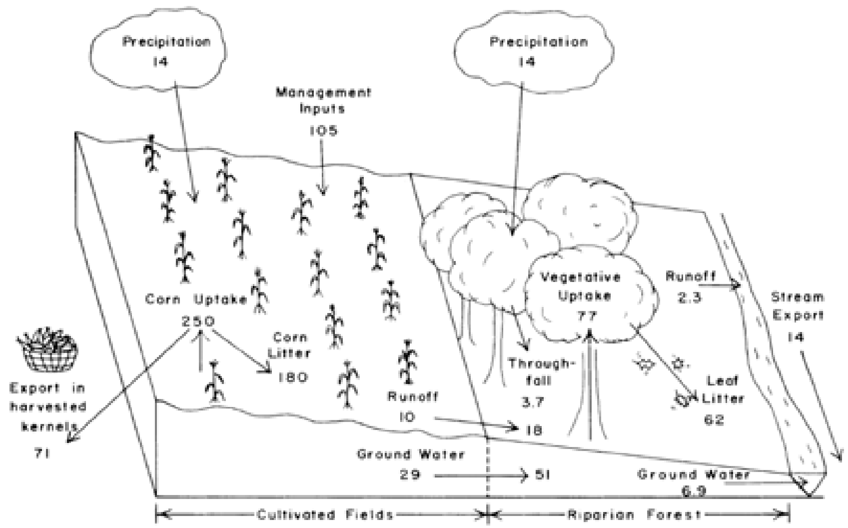


Fig. 2. 18 Total nitrogen flux from cropland to a river via a downslope forest (Peterjohn and Correll (1984 Ecology 65: 1466-1475).

Fig. 2. 19 Summary of the findings on the *downslope utilisation of leached nutrients function* of forest (see Fig. 2.3 for key)

2.4.10 Downslope (and coastal) physical function

Riparian forest strips (sometimes known as ‘buffers’) have been used to trap sediments and any sediment-bound chemicals (e.g., phosphorous or pesticides) being transported in overland flow (Path F or J) from upslope cropland. Peterjohn and Correll (1984) cited within in the last section is a good example of where this can be effective. A more recent example is that of Santos and Sparovek (2011) who demonstrated the value of riparian forests in trapping sediment from upslope cotton farming at a site in central Brazil. In another recent example from Brazil, Bicalho *et al*. (2010) demonstrated that herbicides (i.e., Diuron, Haxazinone and Tebuthiuron) applied to sugar cane crops could be trapped by riparian forest. This water-quality related function has similar water-quality related provisioning, regulating and supporting ecosystem services as the previous downslope function (Section 2.4.9).

The presence of riparian trees can regulate the thermal regime of rivers (see Section 2.4.5). This function affects the regulating service of water quality, and the supporting services of aquatic habitat and biodiversity.

Additionally, closed forests and open woodland within river flood plains are known to reduce the speed of flood flows travelling across flood plains more than lower herbaceous vegetation (Straatsma and Baptist, 2008). This effect reduces the return of over-bank flows back to rivers (thereby mitigating downstream peak flows; Section 2.4.10), and also enhances flood plain infiltration (Section 2.4.6). A similar effect is afforded by mangrove forests that can better attenuate inland flooding by seawater in comparison to lower herbaceous vegetation (Gedan *et al*., 2011). These flood-related physical functions have the specific regulatory ecosystem service of mitigating flood hazard further downstream or further inland, respectively.

As with the *downslope utilisation function* the effectiveness of the sediment trapping function (known as the 'trap efficiency'), is seen to be site dependent whether beneath forest or other land-covers (Ziegler *et al*., 2006). This also probably applies to the flood attenuation potential of forests. Similarly, no systematic global analysis of the extent of the conditions conducive to the water or sediment trap efficiency of riparian forests has been undertaken, though it is known that *riparian and coastal-mangrove wetlands* (see Acreman – this volume) do cover relatively large areas of the globe (Fig. 2.19).

Fig. 2.20 Summary of the findings on the *downslope physical function* of forest (see Fig. 2.3 for key)

2.4.11 Peak-flow damping and low-flow enhancement function

The greater rates of wet-canopy evaporation (Path C: Fig. 2.3) from forests compared to those from herbaceous vegetation occur during rainy periods (Calder, 1990; Kume *et al*., 2008) and so should have a direct damping impact on river peak-flows in forest-covered basins. This effect is however moderated by the observation that catchment-average rates of wet-canopy evaporation (mm/hr) are typically much smaller e.g., one fifth of those of riverflow per unit area (mm/hr). Annual transpiration rates are often comparable to those for annual riverflow per unit area and so may have a larger impact on peak-flow if affected by a change of vegetation. Transpiration losses from catchment systems are however distributed over much longer periods than wet-canopy evaporation (Kume *et al*., 2008), so this may partially negate the beneficial effect on peak-flows inferred from greater long-term rates. The greater evaporation from forests may have an additional indirect impact on peak-flows. Greater evaporation will dry the soil more, and because of the inherent nonlinearities in catchment response (Young and Beven, 1994), this can have a disproportionately large mitigating effect on the rates of lateral subsurface flow in soil strata (Path H: Fig. 2.3) and so reduce peak-flows.

As noted in Section 2.3, infiltration-excess overland flow (Path F) does not produce more than a few percent of the riverflow in most vegetated areas (Dubreuil, 1985). Consequently, an enhancement of the infiltration capacity (Path G) through the planting of trees (Section 2.3), cannot remove any more than the few percent of the riverflow generated by infiltration-excess overland flow, and so cannot significantly effect on the peak-flows in rivers for most areas (see e.g., Chappell *et al*., 2006). Only in localised areas of very slowly permeable topsoil (e.g., FAO Gleysol, FAO Vertisol) that coincide with areas dominated by intense rainfall (e.g., areas below the tracks of tropical cyclones or extreme rainfall events in other areas of the globe), might the effect of trees on infiltration capacity affect river flows. Clear observational evidence of the effect of forests in these localised areas (Zimmerman *et al*., 2012) or during extreme events (e.g., 1 in 100 year rainstorms) is however lacking for most areas with humid climates.

Given these complex interactions, changes in peak-flow as a result of the presence of forests may be best examined by studying their integrated effects on peak-flow following a land-cover manipulation of forest cutting or planting. Guillemette *et al*. (2005) reviewed the impact of forest cutting in forests of boreal and temperate climates. They showed that 74 out of 75 studies showed either no change or an increase in peak flow with forest cutting. Most studies showed a 0 to 100 percent increase in peak-flow and a further four studies up to 170 percent increases (Fig. 2.21).

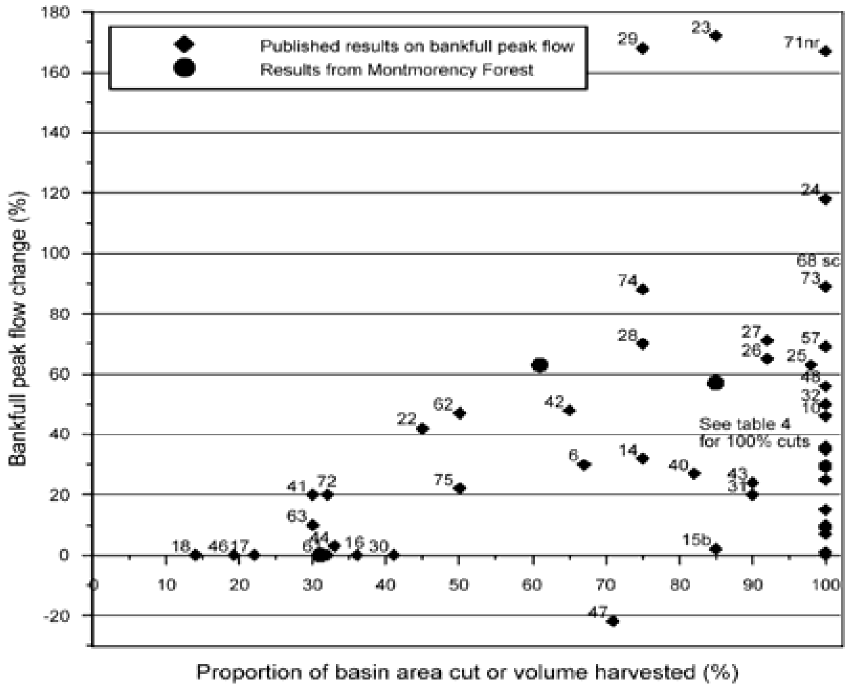


Fig. 2.20 A review of changes in river peak-flow following forest cutting in boreal and temperate regions by Guillemette *et al*. (2005 J. Hydrol. 302: 137-153).

Most studies from tropical climates similarly increases in peak-flow with forest cutting or reductions in peak-flow with forest planting (e.g., Fig. 2.21).

The notable exceptions to this general trend arise where the preparation of wetland sites for plantations involves cutting surface drainage channels, which can add new rapid pathways that can increase peak-flow (e.g., Fig. 2.22).

The dominance of a beneficial (i.e., reducing) effect of forests on peak-flows means that this function should be considered beneficial to the regulating ecosystem service of flood mitigation. However, the present inability to explain the wide range in the mitigation effect means that more work to strengthen the observational evidence is needed (Fig. 2.23). Moreover, all of the findings relate to small basins and should not be extrapolated to the behaviour of large rivers where the effects of channel routing dominate, where trees have reduced ability to mitigate channel velocities.

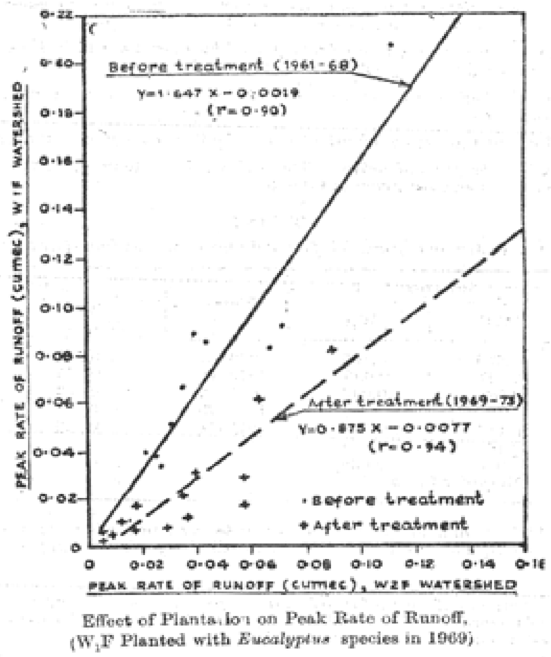


Fig. 2.21 Decreases in river peak-flow following tree planting shown by Mathur *et al*. (1976 Indian Forester 102: 219-226) in tropical India.

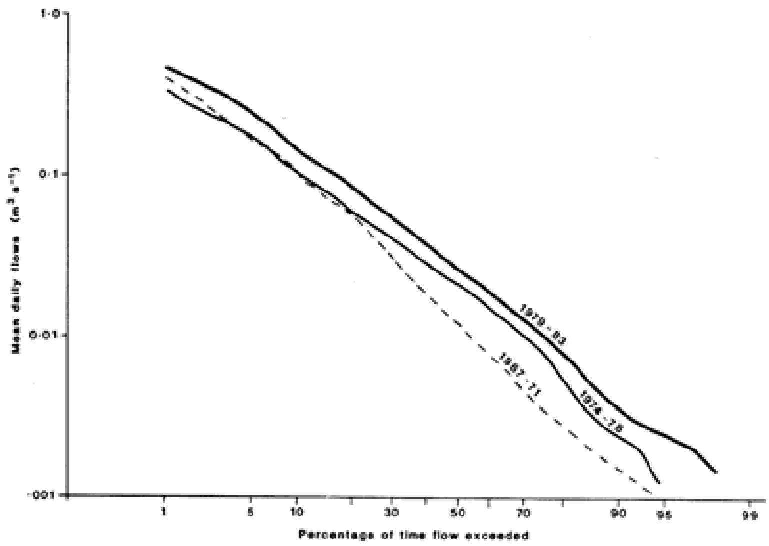


Fig. 2.22 Increases in river flows occurring for only 10 percent of the time (Q10) following the addition of forestry drainage channels to an upland wetland. The broken line is the 'flow duration curve' prior to drainage, and the solid lines the 'flow duration curves' for periods post drainage (Robinson *et al*. 1998 IoH Report 133).

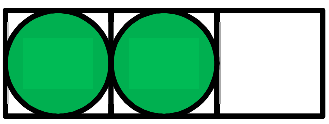


Fig. 2.23 Summary of the findings on the *peak-flow damping function* of forest (see Fig. 2.3 for key)

The observed evidence for the beneficial effects of forests on river low-flows does not match the perception of local farmers within the tropics (Pereira, 1959). Indeed, the review of comparative basin studies within the tropics by Bruijnzeel (1990) showed that forests are more likely to reduce river low-flows and thereby have a negative impact on the provisioning ecosystem service of water supply.

This negative effect could be attributed to the greater rates of transpiration from forest when compared with cleared land.

There may be circumstances where forests can enhance river low-flows. In areas of high rainfall intensity coincident with slowly permeable topsoils (e.g. FAO Gleysols, FAO Vertisols)
a significant proportion (e.g., 50 percent) of the riverflow may be produced by infiltration-excess overland flow (Path F). If a significant proportion of the infiltration-excess overland flow can be diverted into the deep subsurface (Path I) via improvements to infiltration and easy vertical drainage thereafter, then river low-flows might be increased. However, to observed increases in low-flows following tree planting, the change in evaporation (mm/yr) must be a smaller than the change in infiltration-excess overland flow (mm/yr). This is the so called 'infiltration trade-off hypothesis', and clear evidence to support this hypothesis has not yet been collected (Bruijnzeel, 2004). Consequently, robust evidence for the low-flow enhancement effect of forest is not yet available (Fig. 2.24), and further research needed to support this function.

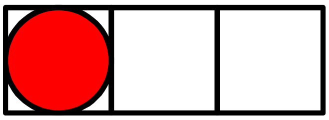


Fig. 2.24 Summary of the findings on the *low-flow enhancement function* of forest (see Fig. 2.3 for key)

2.4.12 Enhancement of river water quality function

Within forests, the *reduced slope erosion function* (Section 2.4.7), the *exclusion of pollutant inputs function* (Section 2.4.8), the *downslope utilisation of leached nutrients function* (Section 2.4.9) and the *downslope physical function* (Section 2.4.10) all mean that natural forests should enhance river water quality. A global assessment of the water quality of rivers only draining natural forest versus those draining cropland, improved pasture or urban areas has yet to be published. Some studies are however available that show the low nutrient contamination (from agricultural fertilisers: Section 2.4.8) of the largely forested headwaters of the Amazon basin relative to other rivers (Figueiredo *et al*., 2010). There are localised exceptions to these findings – certain tree-crop plantations, notably oil palm, have high chemical inputs that may leach (via Paths F, H, I and J) in to rivers (Halimah *et al*., 2010).

Agricultural productivity of croplands is sometimes supported by irrigation with alluvium-rich river-water. Where this process increases the rates of deposition of alluvium on the upstream flood plain, then the natural rates of sedimentation (that includes nutrients) on downstream river deltas may be reduced (in the same way that large dams reduced natural loads of alluvium). Where a forest cover discourages or excludes such irrigation activities it will enhance the provisioning service of downstream fisheries and the supporting services for deltaic habitat maintenance and associated biodiversity.

Overall the expected better water quality of rivers within natural forestlands, particularly due to the *exclusion of pollutant inputs function*, should benefit the provisioning service of clean river-water available for water supply abstractions, but more robust global data are needed to underpin specific policy recommendations (Fig. 2.25). Additionally, maintaining a natural nutrient cycle is supporting ecosystem service i.e., one that is essential for aquatic life.

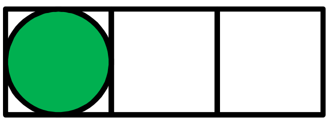


Fig. 2.25 Summary of the findings on the *enhancement of river water quality function* of forest (see Fig. 2.3 for key)

2.5 Related issues: carbon and water cycle interactions

The need to quantify the ability of different types of biome to capture, retain or lose carbon is major global issue (Yuan *et al*., 2009). These ‘carbon pathways’ are closely associated with the hydrological pathways. Whether a land-cover is capturing more carbon dioxide (i.e., downward flux of CO_2) or returning it to the atmosphere (i.e., upward flux) is measured directly from the direction of the vertical wind eddies and the associated concentration in the atmosphere, as is evaporation (Paths C and D). This balance is also affected by the moisture status in the soil (i.e., water within Path H; Cabral *et al*., 2011). The loss of carbon from soils into rivers, where it is then lost to atmosphere as CO_2 (Richley *et al*. 2002) or oceans in dissolved and particulate forms (Neu *et al*., 2011), is dependent on the surface and subsurface hydrological pathways (Paths F, H, I, J and K of Section 2.3).

2.5.1 Carbon dioxide capture function

Current evidence demonstrates that for the same temperate latitude, undisturbed forests capture more CO_2 than does grassland (Fig. 2.26; Valentini, 2007). Forests therefore contribute to the regulating environmental service of better carbon sequestration. However, some boreal deciduous forests and some temperate conifer forests have a net ecosystem exchange that shows they are losing more CO_2 than they are accumulating (Fig. 2.26). Undisturbed tropical forests tend to be accumulating CO_2, though not when they are disturbed and drained (Hirata *et al*. 2008).

2.5.2 Aquatic carbon source function

Very little observed data are available that can illustrate the differential effects of forest, grassland or crop land-covers on the release of dissolved and particulate carbon to rivers, particularly in tropical environments. Richey *et al*. (2002) controversially suggested that CO_2 degassing from rivers in Amazon basin could amount to 1.2 ± 0.3 Mg C / ha / yr, which is equivalent to the CO_2 losses from the forest canopy. As forest disturbance accelerates the loss of carbon to rivers (Schelker *et al*., 2012), the regulating environmental service of better carbon sequestration may apply to undisturbed natural forests. As the carbon naturally added to rivers provides food for the aquatic biota (Nystrom *et al*. 2003), then the maintenance of natural forests, particularly in riparian zones, helps maintain aquatic biodiversity, thereby providing a supporting ecosystem service.

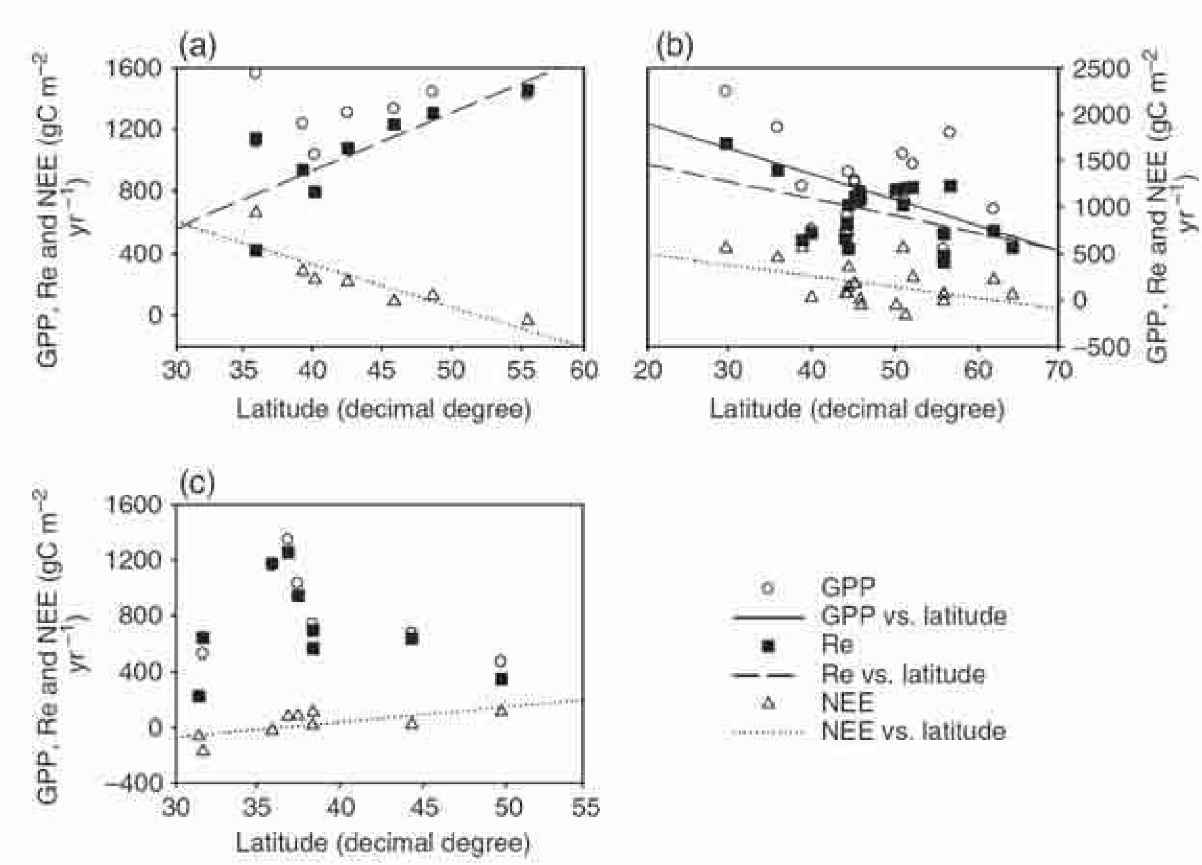


Fig. 2.26 The relationship between latitude and the accumulation of CO_2 (shown with a positive net ecosystem exchange, NEE) for: (a) deciduous forest, (b) conifer forest and (c) grassland biomes (from Yuan *et al*. 2009. Glob. Change Biol. 15: 2905-2920).

2.5.3 Hydrological co-benefits of enhancing the carbon function of forests

Of equal importance when considering carbon and water cycle interactions are the *co-benefits* to hydrological functions that should come from new schemes (e.g. United Nations 'Reducing Emissions from Deforestation and Forest Degradation' or REDD) to retain carbon within the landscape (Section 2.7). If the loss of forest-cover is reduced by REDD or forestry management enhanced under REDD+, then co-benefits to water-related ecosystem services should be produced (Strickler *et al*., 2009). However, scientific investigation is needed to quantify these hydrological co-benefits of carbon sequestration (Section 2.5), not a return to the myths or misinterpretations of the forest-water interactions (Malmer *et al*., 2010).

2.6 Economic values of water-related forest ecosystem services

Ecosystem services are humankind benefits that are supplied by natural ecosystems or natural capital (Kareiva *et al*., 2011). Those services specifically related to water are sometimes called *watershed services* (Stanton *et al*., 2010) or *water services* (Perrot-Maître and Davies, 2001). The assessment of observational evidence quantifying the hydrological functions of forests was presented in Section 2.5. This assessment shows that forests deliver a range of watershed or water-related ecosystem services. Six of the ten hydrological functions did however relate to the value of forests for delivering better quality water within rivers (Sections 2.4.6, 2.4.7, 2.4.8,.2.4.9, 2.4.10 and 2.4.12). This has direct and indirect impacts on the *ability of rivers (and any associated water supply reservoirs) to provide resources clean enough for water supply abstractions* (Shaw *et al*. 2010) Consequently, the regulating service of water quality (Millennium Ecosystem Assessment, 2005) is a fundamental control on the provisioning ecosystem service of water supply, and so the hydrological function of water quantity

(Section 2.4.3) should not be considered in isolation from the hydrological functions related to water quality (see above). This assessment of the hydrological functions of forests has also highlighted that not all management systems within the globe's forests are beneficial to ecosystem services. Forestry (within natural forests or plantations) that involves: (1) drainage, (2) the application of fertilisers and/or pesticides, or (3) intensive logging has a negative impact on various water quality related functions (see above) and peak-flows (Sections 2.4.10). The absence of drainage and chemical additions plus the need for reduced impact forms of timber harvesting (Section 2.7) may be required for the ecosystem service benefits to be seen.

Case studies are available that demonstrate reforestation and improved forest management can so improve the water quality of rivers that the benefits to water supply economics outweigh the costs of the ecosystem service schemes. A good example comes from the temperate forests of the north-eastern USA. In order to improve the river water quality in the Catskill and Delaware catchments for water supply abstractions, the City of New York invested $1.0 to 1.5 billion in improved forest (and agricultural) management, including reforestation. This was financed by a 9 percent tax increase to water bills. Their only alternative was to construct a new raw water treatment plant that would have required a two fold increase in water bills (Perrot-Maître and Davies, 2001; Stanton *et al*., 2010).

To evaluate water treatment costs, the Source Water Protection Committee of the American Water Works Association conducted a survey in 2002 of approximately 40 water suppliers (Fig. 2.27).

% of Watershed Forested	Treatment and Chemical Costs *per mil gal*	% Change in Costs	Average Treatment Costs *per day at 22 mil gal*
10%	$115	19%	$2,530
20%	$93	20%	$2,046
30%	$73	21%	$1,606
40%	$58	21%	$1,276
50%	$46	21%	$1,012
60%	$37	19%	$814

Fig. 2.27 Water treatment (including chemical) costs based on percent of forested water supply catchment (Ernst *et al*., 2004).

Their survey results indicated that for every 10 percent increase in forest cover in the water supply catchments (up to about 60 percent forest cover), treatment costs decreased approximately 20 percent. They also found that 50-55 percent of the variation in the treatment costs could be explained by the percent forest cover in the water supply catchments (Ernst *et al*., 2004). The reasons for the beneficial effect of forests were not explained, though the role of the exclusion of pollutant inputs function of forestlands (Section 2.4.8) must be a significant factor.

Payments for Ecosystem Services (PES) are payments or exchange of credits between a buyer and seller to effect some improvement in the ecosystem service. There is a large potential for these payments to deliver water quality improvements given the current market value of water quality in the global environmental market (Fig. 2.28).

Environmental Market	Market Value (2008)
Regulated Carbon	$117,600,000,000
Water Quality	$9,250,000,000
Biodiversity	$2,900,000,000
Voluntary Carbon	$705,000,000
Forest Carbon	$37,100,000

Sources: World Bank "State and Trends of the Carbon Markets: 2010." Ecosystem Marketplace Reports: "Building Bridges: State of the Voluntary Carbon Markets 2010" and "State of Biodiversity Markets: Offset and Compensation Programs Worldwide".

Fig. 2.28 Market value of environmental markets in 2008 (Stanton *et al*., 2010).

The markets for PES including *Water Quality Trading* (where water quality regulated organisations purchase and trade in offset credits to meet their obligations) are already established across the globe and continue to grow (Fig. 2.29).

	Programs Identified	Active Programs	Transactions 2008 (US$ Million)	Hectares Protected 2008 (million ha)	Historical Transactions through 2008 (US$ Million)	Hectares Protected Historically
Latin America	101	36	31	2.3	177.6	NA
Asia	33	9	1.8	0.1	91	0.2
China*	47	47	7,800	270	40,800	270
Europe	5	1	NA	NA	30	0.03
Africa	20	10	62.7	0.2	570	0.4
United States	10	10	1,350	16.4	8,355	2,970
Total PWS	216	113	9,245	289	50,048	3,240
Water Quality Trading	72	14	10.8	NA	52	NA
Totals	**288**	**127**	**9,256**	**289**	**50,100**	**3,240**

* Note: We separate China from the rest of Asia given the level of activity.

Fig. 2.29 Summary of PES transaction data for 2008 and historically (Stanton *et al*., 2010).

What is currently missing from PES analysis is a systematic economic valuation of each hydrological function of forestlands. Without this it is difficult to accumulate the financial benefits for a specific ecosystem service (e.g., provision of water supply) from the multiple hydrological functions. Equally, it is difficult to estimate the *trade-offs* between the beneficial and negative hydrological functions of forests at a particular location. Research on the economic valuation of each hydrological function of forests is needed.

2.7 Global and regional policies

A central aspect of the Convention on Wetlands ('Ramsar Convention') is the 'conservation and wise use of wetlands', under whatever land-cover, including forests. Given that riverine, lacustrine and palustrine ('bogs') wetlands typically receive their water from a much larger catchment area, then the land-cover on the surrounding catchment is also of fundamental importance to wetland conservation and management.

To achieve this mission, Ramsar recognise that better quantification of the ecosystem services delivered by wetlands is needed (Strategy 1.4ii of the *Ramsar Strategic Plan 2009-2015*) and underpinned by a robust understanding of the science, e.g., hydrological processes and pathways (Strategy 1.6; Section 2.3). Better scientific and financial evidence for wetland services should deliver greater cross-sectoral recognition of the significance of wetlands in decision-making. Quantification of the hydrological functions of forests (whether in or upslope of wetland areas) and the resultant assessment and valuation of ecosystem services delivered, is just one land-cover type associated with wetlands, and needs to be considered with the agricultural, grassland and urban land-covers. There is also an appreciation that many different hydrological functions affect a particular wetland and it may have many different users. Hence there is an appreciation the different functions and user needs must be assessed and managed together within an *Integrated Water Resources Management* approach (Strategy 1.7).

Ramsar is now working more closely with the Convention on Biological Diversity (CBD) to deliver its goals (Strategy 3.1). At the heart of CBD's Strategic Plan are 20 targets to be met by 2020, collectively known as the *Aichi Biodiversity Targets*. Several of these targets directly relate to both wetlands and forests (CBD, 2012). Notably:

Target 11: At least 17 per cent of terrestrial and inland water areas are conserved, and

Target 14: Ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded.

These strategies and targets are compatible with the desire of national and international forestry organisations to improve forestry management to increase the delivery of water-related ecosystem services. Most national forestry departments now have guidelines that seek to minimise the impact of forestry operations and improve the water service delivery. For example, management of the temperate forests (mostly plantations) in the United Kingdom is guided by the 'Forests and Water Guidelines' (Forestry Commission, 2003). Equally, management of the tropical natural forests in the State of Sabah (Malaysia) is guided by the 'RIL Operation Guide Book' (Sabah Forestry Department, 1998).

In forest blocks where forestry operations are subject to independent certification, guidelines can be replaced by very specific environmental criteria that <u>must</u> be met if a certificate of sustainable forest management is to be awarded and maintained thereafter. These certificates are particularly important within regions of tropical natural forests as some of the largest land-cover changes are taking place in this biome (Drigo, 2004), and some of the largest negative hydrological impacts seen (Chappell *et al*., 2004). Very few rigorous studies have attempted to quantify the effect of tropical forest certification on water services. Thang and Chappell (2004) have however shown that one example of such rules (i.e., the Malaysian Criteria and Indicators for forest management certification, or 'MC&I') are at least compatible with the delivery of water services. Forestry management outside of those forest blocks that are closely scrutinised by independent assessors, needs to adopt at least some strict rules (not simply 'guidelines') that are then shown to deliver water services locally (Chappell and Thang, 2007).

Some countries, notably India and China, are following policies of rapid and extensive reforestation with the aim of delivering water-related ecosystem services (Ravindranath and Murthy, 2010). The delivery of such services needs an equal level of scientific investigation and scrutiny by independent assessors, if others (e.g., FAO, 2005; Hayward, 2005; Calder and Aylward, 2006) are not to challenge the stated water services being delivered. Such an objective would be compatible with Strategy 1.6 of the *Ramsar Strategic Plan 2009-2015*, as noted earlier.

2.8 Management options

The management options to enhance the delivery of beneficial ecosystem services via changes to the hydrological functions of forests will be site specific, depending on ‘forest type’, local hydrological conditions and end-user requirements. Many wetlands or catchments have multiple land-covers (forest, cropland, grassland, urban), so the cumulative and net effects of the ecosystem services for each land-cover need to be considered together, thereby paralleling the policy approaches of *Integrated Water Resource Management* (IWRM).

2.9 Policy recommendations for future activities

The policy recommendations jointly to Ramsar and CBD resulting from this assessment of the observed evidence for the *hydrological functions of forestlands* (and the subsequent impact on water-related ecosystem services) are as follows:

1/ All assessments of the water-related ecosystem services from local to international scales will need to be based on sound hydrological science (notably the *dominant hydrological pathways*), and a thorough evidence-based (observational) understanding of the impacts of land-cover and associated management on the *hydrological functions*. This view is compatible with Strategy 1.6 of the *Ramsar Strategic Plan 2009-2015.*

2/ Assessment of the effect of forests (and associated management) on water-related ecosystem services will need to be *balanced*, namely it will need to cover both the physical (e.g., water quantity, river peak-flow, sediment trapping) and water quality related (e.g., exclusion of chemical inputs, soil conservation, nitrate utilisation) functions at local to international scales.

3/ The effect of forests on hydrological functions will need to quantify the effect of the different forms of forestry management, including plantation-related drainage, agro-forestry impacts of livestock or chemical additions, and timber harvesting impacts, where present.

4/ A systematic and in depth global review of hydrological functions of forests related to *water quality effects* will need to be undertaken to provide a clearer evidence base for ecosystem service valuation, management and to convince policy makers of the need to value the water services provided by forests.

5/ An equally rigorous assessment of the impact of forestry certification criteria and forestry management guidelines on hydrological functions will be needed.

6/ Some highly targeted experimental studies (with new observations) will be needed to quantify those hydrological functions of forests with a poor evidence base (sometimes linked to existing forestry management guidelines or rules), and the potential to have a significant global impact, and

7/ A systematic financial assessment of the impact of each forest hydrological function on the value of the ecosystem services delivered, will need to be undertaken; this would provide clearer financial evidence to convince policy makers of the need to value the water services provided by forests.

With a stronger evidence-base, policy makers may be more willing to make the financial investments necessary to deliver greater water services within forest-rich environments.

CHAPTER 3 Wetlands

3.1 Definition

Wetlands occupy the transitional zones between permanently wet and generally drier areas; they share characteristics of both environments yet cannot be classified unambiguously as either fully aquatic or terrestrial (Mitsch and Gosselink, 2007). It is the presence of water for some significant period of time that creates the soil, its micro-organisms and the plant and animal communities, such that the land functions in a different way from either aquatic or dry habitats (Acreman and Mountford, 2009). Wetlands include a range of soils (*e.g.* peat in fens, alluvium in floodplains and marine clays in estuaries), vegetation communities (*e.g.* grasslands, forests, mangroves, reed-beds), animals (*e.g.* fish, reptiles, amphibians) and microbes (*e.g.* methane producing bacteria). Many local terms are applied to wetlands, including such general anglicised terms as 'marsh', 'swamp', 'bog' *etc*, and regionally specific terms such as aapa mires (rheotrophic mires of the boreal zone), billabongs (ox-bow lakes), fadamas (floodplain farmland in Nigeria) and dambos (headwater wetlands in southern Africa).

The international Convention on Wetlands, the intergovernmental treaty established in Ramsar, Iran in 1972 (Ramsar Secretariat, 2011), provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources. The “Ramsar” Convention has been signed by 162 countries and adopts an extremely broad approach and defines wetlands as: '*areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six metres*'. Such a definition thus includes many ecosystems from coral reefs to lakes in underground caves. Many countries have produced variations on this broad definition. For example, Canada defines wetland more specifically as "*land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment*".

3.2 Extent

The amount of wetland globally is difficult to quantify because detailed inventories generally do not exist and definitions of extent and loss are subject to a wide range of interpretations. The UNEP-World Conservation Monitoring Centre has estimated that wetlands cover around 570 million hectares (5.7 million km^2); roughly 6% of the Earth's land surface, which is consistent with past estimates of 6% (Maltby & Turner 1983) and 4 to 6% (Mitsch and Gosselink, 2007). Though the Ramsar Convention (Ramsar Secretariat, 2011) report a 'best' minimum global estimate at between 748 and 778 million hectares.This is probably about half of the extent that existed before human modifications during historical times (Maltby 1986). Many European countries have lost between 50% and 70% of their wetlands in the last century (McCartney *et al*., 2000). For example, France's largest wetland type, the freshwater meadows, bogs and woods, once covered 1.3 million ha but during the 20th century it declined at a rate of 10,000 ha per year (Baldock, 1990). In Greece a 60% loss of wetlands, mainly lakes and marshland, took place by land drainage for agriculture after 1925 (Maltby, 1986). In the same way, 28% of Tunisian wetlands had disappeared in the 20th century. In Asia, about 85% of the 947 sites listed in the Directory of Asian Wetlands were under threat with 50% of the threatened sites being under serious threat (Scott and Poole, 1989). By the modern era, the United States had lost some 54% of their original 87 million hectares of wetlands (Tiner, 1984), primarily to drainage for agricultural production. In Asia some 67% and in Latin America and the Caribbean 50% of the major threats to wetlands were from hydrological change related to drainage for agriculture, pollution, catchment degradation or diversion of water (WCMC, 1992).

The Millennium Ecosystem Assessment (2005) reported that aquatic ecosystems are the most severely impacted ecosystems, particularly from the withdrawal of water for direct human needs for drinking, growing crops and supporting industry, with many impacts directly the result of fragmentation by dams (Nilsson *et al*. 2005). Worldwide, conversion or drainage for agricultural development has been the principal cause of inland wetland loss. The World Commission on Dams

(2000) found that floodplain and deltaic wetlands downstream of dams have been degraded in many parts of the world due to lack of floods.

3.3 Hydrological processes

Wetlands are intrinsically linked to the water cycle, both through the interactions between wetlands and the wider environment and internal hydrological processes. Hydrology is probably the single most important determinant for the establishment and maintenance of specific types of wetlands and wetland processes (Mitsch & Gosselink, 2007). Even when hydrological conditions in wetlands change even slightly, the biota may respond with large changes in species richness and ecosystem productivity. In return, wetlands are significant in altering the water cycle (Bullock and Acreman, 2003), influencing evaporation, river flows, groundwater and lake levels, so although they cover only 6% of the land surface they influence much of the globe.

Wetlands in uplands areas, in the headwaters of river basins, tend to be fed directly by precipitation, with water leaving the wetlands as evaporation, out-flow of surface water or percolation into the ground. Wetlands downstream have the additional input of river water or groundwater, whilst those on the coast will also be subject to tidal inflow and outflow of marine water. Thus the nature of interactions between wetlands and the water cycle is very wide and complex and varies between individual wetland types. In addition to natural hydrological processes, many wetlands are influenced by built infrastructure and water management schemes, including dams, water transfers, abstractions, discharges and pumping. Some of these influences may have negative impacts on wetlands, such as the reducing river flows onto floodplains regime downstream of a dam. In others cases, infrastructure is an integral part of the maintenance of ecological character within a wetland, either to counter external pressures, such as through artificially holding water during wet periods for use in the wetland during dry periods, or because the infrastructure has created the wetland, for example some reservoirs behind dams have become ecologically important wetlands, such as Rutland Water in the UK.

Water quality will also be determined by the interaction between the wetland and its environment and the hydro-chemical processes within the wetland itself. Physical hydrological processes have implications for water quality. For example, wetlands fed exclusively by rainfall such as upland blanket bogs, may be quite acidic, whilst those fed by chalk or limestone aquifers, such as fens, may have basic waters. Relative amounts of more than one water source lead to subtle chemical balance that can produce unique habitats. Salinity will be controlled by interactions with the marine environment or through evaporation in endorheic wetlands, such as soda lakes including Lake Nakuru in Kenya. Wetlands are vulnerable to pollutants carried by incoming waters including nutrients (e.g. nitrogen and phosphorus) - which although are essential to life can have negative effects in large quantities - heavy metals (copper and zinc), toxic pharmaceuticals (e.g. cytotoxics and anaesthetics) and sediment that may be inert or have others pollutants bound to it.

3.4 Evidence for hydrological functions

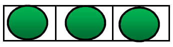
Most wetland literature refers to hydrological functions of wetlands. In particular, wetlands are reported to "act like a sponge" (an analogy which goes back as least as far as Turner, 1757, p30), soaking-up water during wet periods and releasing it during dry periods (e.g. Bucher *et al*, 1993), thus reducing floods and preventing droughts. Wetlands are also reported to be "the kidneys of the landscape" because they cleanse polluted waters (Mitsch & Gosselink 2007). The case for wetland conservation is often made in terms of ecosystem services including groundwater recharge and discharge, flood flow alteration, sediment stabilization, water quality (Maltby 1991). This has been promoted as a potential means of catchment management by organisations such as IUCN (Dugan, 1990), Wetlands International (Davis and Claridge, 1993) and the Ramsar Convention on Wetlands of International Importance (Davis, 1993). They have influenced international wetland policy (OECD, 1996) and its uptake at the national (e.g. South Africa, Zimbabwe and Uganda), and continental levels e.g. Europe (CEC, 1995; Blackwell and Maltby, 2006) and Asia (Howe *et al*, 1992). Despite these policy initiatives, scientific evidence for the role of wetlands in the water cycle is difficult to find and often confusing, if not contradictory. The terms wetlands covers many land types and each type functions hydrologically in a subtly different way, thus, whilst there are clear examples of some services, it is difficult to make generalisations for all wetlands.

Bullock and Acreman (2003) reviewed over 400 papers on wetlands published during the period 1930-2002 and produce a database of 169 scientific studies that reported quantitative new findings, totalling 439 statements on the hydrological significance of wetlands. Of the 439 statements, only 83 (19%) conclude the wetland influence on a single hydrological measure to be neutral, insignificant or no to occur. The vast majority conclude that wetlands either increase or decrease a particular component of the water cycle. Most, but not all, studies (23 of 28) show that floodplain wetlands reduce or delay floods, with examples from all regions of the world. This same flood reduction is also seen, but less conclusively (30 of 66) for wetlands in the headwaters of river systems; a substantial number (27 of 66) of headwater wetlands increase flood peaks (these studies were mostly from Europe, but included work from West Africa and Southern Africa). Around half of the statements (11 of 20 for flood event volumes and 8 of 13 for wet period flows) show that headwater wetlands increase the immediate response of rivers to rainfall, generating higher volumes of flood flow, even if the peak flow is not increased. The coverage of these studies is worldwide including Africa and South America. There was strong evidence that wetlands evaporate more water that other land types, such as forests, savannah grassland or arable land. Almost two thirds of studies (48 of 74) concluded that wetlands increase average annual evaporation or reduce average annual river flow. About 10% of studies (7) showed the opposite; for example some woodlands in Zambia had greater evaporation than the adjacent wetlands. The remaining 25% were neutral. There was no obvious distinction amongst different wetland sub-types or geographical regions of the world. Two thirds of studies (47 of 71) conclude that wetlands reduce the flow of water in downstream rivers during dry periods. Evidence is mainly from North America and Europe, but includes floodplains in Sierra Leone and wetlands in Southern Africa. This is backed by overwhelming evidence (22 of 23 studies) that shows evaporation from wetlands to be higher than from non-wetland portions of the catchment during dry periods. There is no discernible difference for different wetland sub-types. In 20% of cases, wetlands increase river flows during the dry season. The database contains 69 statements on groundwater recharge; 32 conclude merely that recharge takes place, and 18 conclude there is no recharge. There are similar numbers of studies that report wetlands either to recharge more (6) or less than (9) other land types. Some wetlands, such as floodplains in India and West Africa on sandy soils, recharge groundwater when flooded. Many wetlands have formed at springs and are fed by groundwater. The direction of water movement between the wetland and the ground may change in the same wetland, such as in some peatlands in Madagascar, according to hydrological conditions.

3.4.1 Flood reduction by floodplains

The most compelling scientific evidence is the reduction in flood peaks and delay in flood arrival time downstream of floodplains. Modelling of the River Cherwell, UK (Acreman *et al.*, 2003) showed that removal of embankments, separating the river from its floodplain, would result in a reduction in downstream flood magnitude of 132% due to storage of water on the floodplain and slowing of water speed by friction. In India flood volumes were found to decrease downstream of wide sandy floodplains (Nielsen *et al*. 1991) due to large retention and infiltration losses. Rough vegetation, such as trees and shrubs provide more friction than short grass (Sun *et al*., 2010) and thus increase flood attenuation. Floodplains have been used to manage floods on the large rivers of around the world including the Mississippi (Bedinger 1981) and Rhine (Baptist *et al* 2004). A substantial area of wetland is required to make a significant difference downstream; storage on 3500 ha of floodplain on the River Shannon, Ireland, was equivalent to one day of peak discharge at around $400 m^3 s^{-1}$ (Hooijer 1996). Some of 3,800 hectares of floodplain storage were required on the Charles River, USA to reduce flood risk, whereas the Cherwell floodplains in UK are around 1000 ha. The precise area required will depend on the characteristic of the floods (e.g. volume of flood water, rate of peak flow) to be managed and degree of attenuation desired.

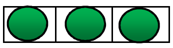
Score



3.4.2 Coastal wetlands and floods

On the coast, mangroves and to a lesser extent salt marshes, can reduce the energy of waves and currents and reduce flood risk from storm surges, stabilising sediment with their roots. It is estimated that for every 5.5 km of healthy coastal wetlands a storm surge travels over, the surge is diminished by 0.3 m (http://healthygulf.org). For example, it was notable that coasts with mangroves were less damaged during the Asian tsunami in December 2004 than those where these wetlands had been removed. By 2080, it is projected that 5-20% of coastal wetlands will be lost due to sea-level rise (Nicholls, 2004). In the Gulf coast area of USA, barrier islands, shoals, marshes, forested wetlands and other features of the coastal landscape provide a significant and sustainable buffer from wind wave action and storm surge generated by tropical storms and hurricanes (Working Group for Post-Hurricane Planning for the Louisiana Coast). Following flooding of the Gulf Coast from hurricane Katrina, restoration of wetlands and barrier islands to help protect New Orleans is being considered as a coastal management option.

Score



3.4.3 Headwater wetlands and floods

The evidence for flood reduction by headwater wetlands is less consistent. When these wetlands are dry or have hollows that can store water, floods may be reduced. However, studies of UK peatlands (Holden and Burt 2003a) showed that the water table was within 40 cm of the surface for 80% of the year and when it rains there is little space for water storage, so most of the rainfall flowed over the peat surface quickly into the river. Much of understanding of flood generation comes from hydrological studies in the USA in 1960s and 1970s. For example, headwater areas permanently or frequently saturated in hollows, at the foot of slopes or flat land adjacent to rivers are called 'contributing areas' (Hewlett and Hibbert 1967). Studies of other headwater wetlands have shown similar characteristics; dambos in Zimbabwe have a small capacity to absorb rainfall at the start of the wetland season, when water table levels are low, but soon became saturated and contributed to flood runoff thereafter (McCartney 2000). Management of upland wetlands can have significant of hydrological functioning. For example removing vegetation can increase the speed of water flow across the wetland surface, such that bare peat could increase the flood peak by between 2 and 19 % (Bonn *et al*., 2009). Drainage has been reported to both increase and decrease flood peaks from wetlands (Holden *et al*., 2004). For example, Burke (1968; 1975) recorded much higher peak flows from un-drained peat areas than from drained areas, Kloet (1971) found that peak flow were increased by drainage, whilst Moklyak *et al.* (1975) concluded that drainage does not always affect the maximum discharge, but may either decrease or increase it. Drains alter hydrological process in two ways. First, drainage can increase water storage capacity within the wetland reducing peak flows and increasing lag times. Second, drainage can provide channels for rapid and direct flow to the stream; this may increase peak flows in the stream. The net result on flood peaks may depend on the type, condition, density and orientation of drainage (Holden *et al.* 2004).

Score

🟢	🟢	🟢	Wetlands with significant surface storage and/or light drainage
🟡	🟡	🟡	Wetlands with little surface storage and no drainage
🔴	🔴	🔴	Wetlands with no surface storage and/or deep drainage/gullies

3.4.4 Wetlands and groundwater

Many wetlands exist because of underlying impermeable layers that prevent vertical movement of water. However, other wetlands are hydrologically connected to underlying aquifers, such as the Azraq oasis in Jordon (Fariz & Hatough-Bouran 1998) which is fed by upward moving groundwater (discharge). During inundation of the floodplain wetlands of the Senegal River floodplain (Hollis 1996) and Hadejia-Nguru wetlands (Hollis *et al* 1993) water moves downwards to the underlying aquifer (recharge). Likewise inundation of the Kairouan floodplain (Acreman, 2000) and Garaet Haouaria marshes (SCET, 1962) in Tunisia recharge the regional aquifer. Alteration of catchment hydrology can have a significant impact on wetlands and the functions they perform. For example, Las Tablas de Daimiel wetlands in Spain (Llamas 1989) was historically groundwater-fed, from water moving upwards from the underlying aquifer. From 1972 irrigated agriculture expanded rapidly and abstraction lowered groundwater levels, reversing the function of the wetland to become a groundwater recharge site.

Managed aquifer recharge is practised widely in India (CGWB, 2005) where millions of structures capture monsoonal rainfall on the surface and allow it to infiltrate into the often low storage capacity basement aquifers. In the Shiquma scheme, north of the Gaza Strip, a small dam has been constructed to create a reservoir which holds flood water. The water is then pumped to large depressions (infiltration basins) in the sand dunes near the coast where it percolates into the ground to recharge the dune aquifer.

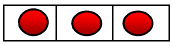
Score

🟢	🟢	

3.4.5 Evaporation

In general, evaporation from ponds and lakes exceeds that from soils, forests and dry grasslands. Wetlands are often composed of areas of open water with emerging large-leaved vegetation providing potential for high evaporation rates from the water surface and through plant leaves. Evaporation rates exceeding 10 mm/day were reported for beds of *Phramites* reeds in northern Germany (Herbst & Kappen 1999), 5 to 12 mm/day Hamun Wetlands in southeast of Iran (Arasteh & Tajrishy 2006) and 4.1-4.9 mm/day for Sudd (Mohammed *et al*. 2008). Linacre (1976) suggested that the presence of swamp vegetation has a relatively minor influences on evaporation rates, whilst Idso (1981) concluded that the presence of vegetation on wetlands does not increase the evaporative loss above that of open water. However, Ingram reported that hydrophytic vegetation can evaporate up to 40% more than open water, rising temporarily to 2.5 in summer. Crundwell (1986) reports wetland evaporation figures of 4 times open water, such as for water hyacinths (Rodgers and Davis 1972). Values of 1.1 are typical for wetlands, such as the Florida Everglades (Allen, 1998). Wetlands in surrounded in arid and semi-arid regions may have particularly high evaporation rates because of energy coming in from the surrounding dry land (called the oasis effect). Most people regard evaporation as a loss of water, yet water is recycled on local and regional scales. For example, the extent of cloud formation and amount of rainfall in arid lands around the Inner Niger Delta in Mali is augmented as the area of wetland inundation increases due to higher evaporation (Taylor 2010); the same may be true for other large wetlands in arid areas such as the Sudd and Okavango.

Score



3.4.6 Water quality improvement

Water flowing through wetlands is often slow due to low gradient and the friction created by dense vegetation. This encourages sediments and other particles that are carried by fast flowing water to settle, it also time for chemical reactions to take place, such as converting harmful substances to an inert form. Because toxicants (like pesticides) often adhere to suspended matter, sediment trapping frequently results in water quality improvements. Water quality can also be improved by the ability of wetlands to strip nutrients (nitrogen and phosphorus) from water flowing through them. It has been reported that wetlands reduce nutrients by encouraging sedimentation (Karr and Schlosser, 1978; Johnston *et al.*, 1984), sorbing nutrients to sediments (see Khalid *et al.*, 1977), taking-up nutrients in plant biomass (Lee *et al.*, 1975) and enhancing denitrification (Lowrance *et al.*, 1984). The Nakivubo Swamp on the edge of Lake Victoria in Uganda receives waste water and untreated sewage from Kampala (Kansiime and Nalubega, 1999) and as a buffer, removing nutrients and pollutants so that the intake for Kampala's water supply is only 3 km away downstream. Soluble phosphorus reaching the River Torridge, UK, from agricultural land was, on average, 73 per cent lower than predicted because of removal by riparian wetlands (Russel and Maltby, 1995). Pollutant removal by natural wetlands has led to wetlands being managed or constructed to act as buffers or for treatment of domestic or industrial waste (Allinson *et al.*, 2000).

Fisher and Acreman (2003) examined results of studies of 57 wetlands from around the world to assess whether wetlands affect nutrient loading of waters draining through them, and showed that the majority of wetlands reduced nutrient loading (nitrogen or phosphorus). Some wetlands however, can increase nutrient loadings for short periods by releasing soluble N and P species, thus potentially driving eutrophication in the receiving water body. N removal is known to be greater in more waterlogged soils (Jordan *et al.*, 1993). The most effective wetlands for water quality improvement are those in which the water flows through the wetland (on-line), such as river entering directly into a swamp. Wetlands that are adjacent (off-line) to the water course, such as floodplains can be very effective at retaining P (Craft and Casey 2000), they only receive a small quantity of the river flow and so have limit effect on the overall water quality.

A major issue is that wetlands can easily become overloaded if pollutant levels exceed critical thresholds, resulting in loss of the toxicant retention services and potentially significant changes to ecological character. However, there are few published records of these threshold levels. N removal efficiency is not affected by the length of time the wetland has received N pollution, while the ability of a wetland to remove P, in contrast, is known to decline with time (Nichols, 1983; Richardson, 1985). Wetlands constructed to remove water-borne phosphate in the UK (Palmer-Felgate, 2009) were found to be effective for the first few years, by locking the nutrients into the sediment. However, the sediments were soon higher in phosphorous than the water entering the wetland and thus were enriching the water. This suggested the need for removing the sediment to revitalise the nutrient absorption process.

Score

			New and managed wetlands
			Old and unmanaged wetlands

3.5 Economic values of water-related wetlands services

The regulating role that wetlands play in the water cycle is very important to people and economists have tried to assess their value in monetary terms (Emerton & Bos, 2004). These include the costs of flood damage that would have occurred if the wetland had not alleviated flood risk and the costs of water treatment works that would need to be built to replace wetland purification services if they were lost. Along the Charles River in Massachusetts, wetlands are utilized in preventing flood damage it was calculated that loss of all wetlands in the Charles River watershed would have caused an average annual flood damage cost of $17 million (U.S. Army Corps of Engineers) concluding that conserving wetlands was a natural, less expensive solution to controlling flooding than the construction of embankments and dams alone, and they proceeded to acquire 3279 ha of wetlands in the Charles River basin for flood protection. If the Lower Shire wetlands in Malawi and Mozambique and the Barotse floodplain in Zambia did not reduce flood risk the costs of damaged roads and houses, relocation of people and loss of farmland would be around US$2 million (Turpie *et al*., 1999). Likewise, natural wetlands in the Zambezi basin, southern Africa, have a net present value of more than US$ 3 million as they reduce flood-related damage costs and worth US$16 million, they also recharge groundwater and purify water to an estimated US$ 45 million (Turpie *et al*., 1999). The infrastructure required to achieve the same level of wastewater treatment to that provided by the Nakivubo swamp in Uganda would cost up to US$ 2 million per year through constructed sewerage and treatment facilities (Emerton *et al*., 1999). The Martebo mire in Sweden maintenance of water quality costs of between US$ 350,000 and US$1 million (Gren *et al*., 1994).

The alteration of the hydrological by wetlands also provides many additional benefits such as through the provision of fish breeding habitat and grazing land on floodplains. For example the economic value of the Senegal River floodplain in west Africa has been estimated (Acreman, 2003) at US$56-136 per hectare for flood recession agriculture, US$140 per hectare for fishing and US$70 per hectare for grazing. In the Hadejia Jama'are floodplain, Nigeria, ecological and social benefits from water in the floodplain were valued at US$9,600 to 14,500 per cubic metre whereas diversion of the water for irrigation was estimated to produce a return of merely US$26 – 40 per cubic metre (Barbier & Thompson 1998)

In addition, the role of wetlands in water cycle is important for social and ecological reasons that are not easy to value economically. For example, annual inundation of floodplains in west Africa and the its implications for grazing, fishing and agriculture are embedded within the social calendar of local people and celebrated with many festivals and social events.

3.6 Related issues

3.6.1 Wetlands and carbon sequestration

Most soils contain some carbon, but peat is particularly rich in carbon. Peat forms when plant material is inhibited from decaying fully by acidic and anaerobic conditions, thus the water cycle is intrinsically-linked to peat formation and conservation. Peat wetlands are found in at least 175 countries, from the tropics to the poles, and cover around 4 million km^2 or 3% of the world's land area. Semi-natural and undamaged peatlands can accumulate carbon at a rate of 30-70 tonnes of carbon per km2 per year (Billett *et al*. 2010; Worrall *et al*. 2010b) now contain 400-700 Gt of carbon in their soils, are at risk if temperatures and rainfall patterns alter with climate change. Peat soils contain a third of the world's total soil carbon. The transfer of carbon between peatlands and the atmosphere as the greenhouse gases carbon dioxide (CO_2) and methane (CH_4) is important for regulating the global climate (Schulze *et al*. 2009). The wetland carbon pool is estimated at 37% of the 1943 Gt of carbon in the terrestrial biosphere pools (Bolin and Sukumar 2000).

Much of the research on carbon budgets has been in northern peatlands. They have over time stored around 450 billion metric tons (or 450 Gt) of carbon in their soils (Gorham, 1991; Maltby and Immirzi, 1993). This is approximately 33 per cent of the global soil carbon stock and is equivalent to 75 per cent of the pre-industrial mass of carbon in the atmosphere and is indicative of northern wetlands historical ability to accumulate carbon. Flux estimates vary from an uptake of more than 220 g CO_2 m^{-2} yr^{-1} to losses of 310 g CO_2 m^{-2} yr^{-1} have been recorded. This focus on northern peatlands tends to mask the large uptake of CO_2 that can occur in tropical peatlands where rates of uptake are in the region of 1800 g CO_2 m^{-2} yr^{-1} but they cover an area of only about 8% of the world's peatlands. Globally, northern peatlands produce between 0.03 and 0.05 Gt tonnes of CH_4 each year (Fung *et al*., 1991; Bartlett and Harriss, 1992). Balancing this, Roulet (2000) estimates from a series of northern peatland studies in Canada, USA and Europe that peatlands are currently a sink of between 20 and 30 g C m^{-2} yr^{-1} (0.104 to 0.156 Gt C yr^{-1}).

The fact that wetlands have accumulated carbon in peat in the past suggests that they may offer a potential climate change mitigation option for the future. However, a review of available European carbon budget data (Byrne *et al*. 2004) concluded that most peatland types vary between a small sink and a moderate source of GHGs, principally from a substantial CH_4 emission; none show unambiguous net uptake of GHG, thus even undamaged peatlands may have a net warming effect on climate, although restored fens and bogs have a much smaller effect than that for various types of pre-restoration management. Therefore, restoration has clear benefits in global warming terms over the un-restored case, even though restored peatlands may not have a net carbon sink function.

Most assimilation and release of carbon occurs in shallow water. The Caspian Sea, for example, is counted as a wetland of some 400,000 km^2, but it is probable that only its coastal edges, amounting to no more than 2000 km^2 in area will be actively absorbing or releasing GHGs (in addition to the normal uptake of CO_2 by the open sea surface). In contrast, the near continuous relatively shallow wetland that covers 160,000 km^2 of western Siberia can be counted as being totally active in absorbing or releasing GHGs.

Interest in the capacity of peatlands to help mitigate climate change through carbon sequestration has stimulated significant questions regarding the status of current peat resources and the possibility of new formation (Immirzi & Maltby 1983; Maltby 2010). This is the case with the potential benefits associated with the blocking of ditches (grips) in the peat to help restore hydrological integrity previously disrupted by drainage. It is still uncertain whether such management actions to increase waterlogging are sufficient to reverse the carbon balance in favour of increased storage (e.g. Worral *et al.* 2003; Worral & Evans, 2009). Other factors such as burning and grazing are important in determining the stability of the existing carbon store. Of overriding importance, however, is whether the current (or immediate future) climatic conditions are sufficient in combination with the local factors such as topography, substrate conditions, vegetation, acidity and nutrient status to enable new peat formation. Despite considerable on-going research there is still uncertainty regarding the existence of the necessary climate template, at least in the UK, for the net accumulation of new peat. It does not detract, however, from the argument to maintain or restore hydrological conditions so as to minimise any further losses of existing carbon store in peatlands.

3.6.2 Wetlands and climate change

The spatial distribution of the world's non-ocean ecosystems is largely determined by two environmental variables: temperature and precipitation (Walter and Breckle, 1985). Climate change is likely to lead to changes in wetlands through direct impacts on precipitation and indirect impacts on evaporation (through changes to temperature and other variables, such as radiation and wind-speed). Winter (2000) undertook a hypothetical assessment of wetlands in different hydrological and landscape settings and concluded that wetlands whose hydrology is dependent on precipitation are more vulnerable to climate change than those fed by groundwater. Burkett and Kusler (2000) recognised that not only is climate change likely to lead to loss of wetlands, such as tundra, marshes and wet meadows underlain by permafrost, but wetlands that are dried can become net sources carbon dioxide (but with possible reduction of methane) serving as a positive feedback to global warming. Parish *et al.* (2007) came to similar conclusions for peatlands. Clément and Aidoud (2007) reported that oligotrophic habitats were the most sensitive palustrian wetlands to climate change. Turetsky *et al.* (2007) found that organic matter accumulation was generally greater in unfrozen bogs compared with permafrost landforms, inferring that surface permafrost inhibits peat accumulation. The diversity of reported response of wetlands to climate change illustrates that such a response is in reality a result of a balance between changes in water table, temperature, nutrient cycling, physiological acclimation and community reorganization (Oechel *et al*. 2000). Kont *et al*. (2007) reported climatically-induced changes in water levels in an Estonian inland bog over a 47 year period. Groundwater levels were found to be rising in a ridge-pool microtope, but falling in a ridge-hollow microtope, demonstrating the potential complexity of wetland response to climate change. Johnson *et al.* (2005) modelled water table levels and vegetation in Prairie wetlands and found that climate change would result in a shift in the productive habitat for breeding waterfowl. In a study of British wetlands, Acreman *et al*. (2009) found that projections of reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities in late summer and autumn with greater impacts in the south and east. In addition, impacts on rain-fed wetlands will be greater than on those dominated by river inflows, whereas minimum water table levels in some groundwater-fed wetlands may be higher.

3.7 Global and regional policies

The Ramsar Convention on Wetlands of International Importance has been signed by 160 countries. 1967 wetlands have been designated as Ramsar sites covering 191 million ha. At the centre of the Ramsar philosophy is the “wise use” concept. The wise use of wetlands is defined as the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development. In turn ecological character includes the services to mankind such as regulation of the water cycle. The Convention on Biological Diversity is the primary instrument at a global scale for the protection of biodiversity is the Convention on Biological Diversity (CBD, 2001), which has at its heart the ecosystem management approach *i.e*. maintaining ecosystem functioning.

At a continental scale, the European Union has several pieces of legislation that protect wetlands and their functions. The Habitats Directive (Directive No 92/43/CEE) was designed to protect the most seriously threatened habitats and species across Europe. This complements the “Birds Directive” (No 79/409/EEC), specifically aimed at avian conservation. At the heart of these Directives is the creation of the network of sites called Natura 2000 comprising: a) Special Protection Areas (SPAs) established under the Birds Directive; and b) Special Areas of Conservation (SACs). The EU Water Framework Directive (2000/60/EC) requires member states to achieve “Good Status” (GS) in all surface and ground waters by 2015. Good Status is defined as slight deviation from the reference conditions, based on populations and communities of fish, macro-invertebrates, macrophytes and phytobenthos, and phytoplankton. Wetlands are explicitly included in the Directive as groundwater dependent terrestrial ecosystems (wetlands connected directly to groundwater bodies) whilst floodplains are considered as part of the surface water body (normally a river) to which they are connected. The North American Wetlands Conservation Act (103 Stat. 1968; 16 U.S.C. 4401-4412) provides funding and administrative direction for implementation of the North American Waterfowl Management Plan and the Tripartite Agreement on wetlands between Canada, U.S. and Mexico.

National policies and legislation highlight the protection of wetlands because of their role in the water cycle. The 1998 National Water Act of South Africa had significant implications for the protection and management of wetlands (Rowlston & Palmer 2002). It recognized riverine, wetland, estuarine and groundwater ecosystems, which must be protected in order to ensure maintenance of the desired goods and services which water resources can provide. Tanzania’s National Water Policy of 2002 follows a similar approach (Ministry of Water and Livestock Development 2002). It includes among its objectives the improved management of ecosystems and wetlands, integrated planning and management of water resources, environmental flows, and the need for these in order to maintain riparian biodiversity, wetland systems and aquatic life. Water is first allocated to basic needs, followed by the environment and then the economy. In the USA, Section 404 of the Clean Water Act (CWA) protects wetlands by regulating the discharge of dredged or fill material into waters. Activities in waters include fill for development, water resource projects (such as dams and levees), infrastructure development (such as highways and airports) and mining projects.

3.8 Management options

Wetlands have some potential used as natural infrastructure options. However, they are not a panacea for flood reduction and pollutant removal. All wetlands are different and each works in a unique manner. Detailed understanding of individual wetlands is essential to understand their functioning and the services they might provide. Often long datasets and detailed process measurement is required, although guidance on rapid assessment of wetland functions is available (Maltby 2009).

Floodplain wetlands store floodwater on their surface and reduce flow speed. Floodplains can be used as ‘washlands’ or flood retention areas upstream of flood risk areas, such as low lying towns. Large areas of floodplain are required to reduce floods significantly, but these can be used for other purposes, such as grazing land or recreation areas at times of no floods. Retention of water may be augmented by building sluice gates and embankments. The fact that few studies suggest floodplain storage will increase floods means that there is unlikely to be a negative impact (a ‘no regrets’ situation).

The utility of upland/headwater wetlands for flood reduction is less ubiquitous. They may reduced or augment floods depending on their characteristics. Restoring and maintaining wetlands with surface hollows may be more likely to reduce floods. If the wetlands are dry before heavy rainfall they are more like to reduce floods; if they are wet they are more likely to generate floods. The uncertainty in whether headwater wetlands reduce floods make employing them in flood management strategies a more risk course of action as they make have unintended consequences.

Restoring vegetation cover to wetland surface will slow water flow velocity on floodplains and headwater wetlands. Planting diverse vegetation, trees, shrubs, tall grasses will increase roughness and reduce floods. Blocking artificial drainage channels in headwater wetlands may reduce floods downstream under certain circumstances.

Wetlands can be effective at removing nutrients, such as nitrogen and phosphorus. They can therefore be useful for treating pollutants. However there are thresholds of pollutant level above which the wetland will be degraded or destroyed, so there are limits to the purification service they can offer. Furthermore, their ability to process pollutant may reduce over time unless the wetland is managed. Management of the wetlands, such as removing sediment saturated in pollutant may be necessary to achieve long term utility.

Wetlands do evaporate as much or more water than other land types. There is little that can be done to reduce evaporation, although minimising open water areas would contribute. However, water evaporated by wetlands is not wasted. Water use is essential for plant growth, which provides habitat and grazing. Water evaporated may not be lost, but may generate local rainfall.

Wetlands overlying aquifers may help to recharge groundwater resources. Available water, such as during floods, may be pumped onto the surface of these wetlands to augment recharge.

3.9 Policy recommendations

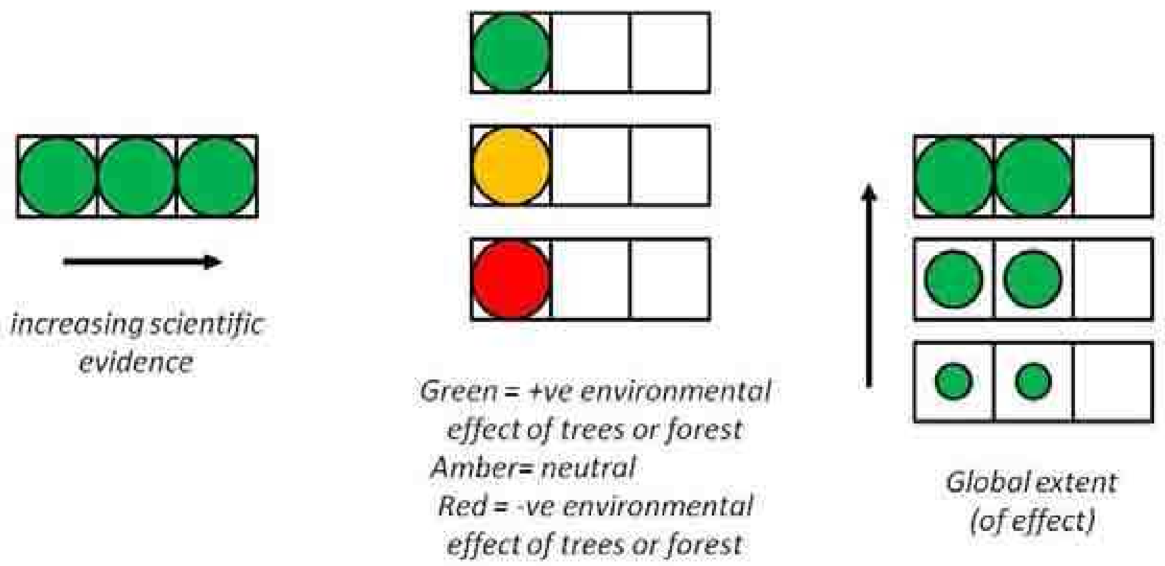
- Floodplain wetlands store floodwater on their surface and reduce flow speed. Large areas of floodplain are required to reduce floods significantly, but these can be used for other purposes, such as grazing land or recreation areas at times of no floods.
- The presence of upland/headwater wetlands may reduce or augment floods depending on their characteristics, such as presence of surface hollows. If the wetlands are dry before heavy rainfall they are more like to reduce floods; if they are wet they are more likely to generate floods.
- Vegetation on a wetland surface will slow water flow velocity on floodplains and headwater wetlands. Diverse vegetation, trees, shrubs, tall grasses will increase roughness and reduce floods.
- Blocking artificial drainage channels in headwater wetlands may reduce floods downstream under certain circumstances.
- Wetlands can be effective at removing nutrients, such as nitrogen and phosphorus, however their ability may reduce over time unless the wetland is managed and there are thresholds above which nutrient removal stops.
- Wetlands do evaporate as much or more water than other land types. However, water evaporated by wetlands is not wasted. Water use is essential for plant growth, which provides habitat and grazing. Water evaporated may not be lost, but may generate local rainfall.
- Wetlands overlying aquifers may help to recharge groundwater resources.

3.10 Gaps

There have been a large number of studies of hydrological functioning of wetlands published in scientific journals and reports. Many are comparisons of river basins with and without wetlands, which means that results are confounded by other differences between the basins. The major gap is the lack of controlled experiments that provide a control on other factors.

The main inconsistency in the literature concerns flood enhancement or reduction by headwater wetlands. More studies are required on the impacts of drainage and blocking of old drains.

Scoring system



CHAPTER 4 Mountains

4.1 Scope of the assessment

Mountains, where virtually all the world's major rivers originate, play a central role in the global hydrological cycle. This section provides a brief review of hydrological processes associated with mountain and the role of mountain ecosystem in the water cycle. Although there is no formal definition of mountains, a commonly adopted definition is based on altitude and slope criteria (Kollmair *et al*. 2005). A more recently proposed definition of mountains suggests using a common ruggedness threshold as a proxy for steepness instead of the more conventional low elevation thresholds (Koerner *et al*. 2011). At a more general level, mountain environments have been defined by features such as limited accessibility, high degree of fragility, marginality, and diversity (Jodha 2000). In this section mountains have been taken as a natural elevation of the earth surface rising more or less abruptly from the surrounding level and attaining an altitude which, relatively to the adjacent elevation, is impressive or notable (Kapos *et al*. 2000).

Natural infrastructure are comprised by biota (animals, plants and other organisms) and their abiotic environment in slow flowing surface waters like lakes, man-made reservoirs or wetlands, in fast flowing surface waters like rivers and creeks, and in the groundwater. In the mountain context natural infrastructure is exceptionally rich in terms of biodiversity (Grabherr and Messerli 2010) and range from tropical and subtropical to temperate climate such as the Himalayas and the tropical Andes. They provide a vast array of services that impact the water cycle including modulating and maintaining climate, provisioning of water, flood control, soil and groundwater, and erosion prevention. Due to the isolated nature and high variability at small scale, mountain biodiversity is highly endemic and vulnerable to climatic and environmental changes from which they typically only slowly recover if at all. In order to conserve mountain biodiversity, around 11 % of all mountain areas accounting for an area of 4.5 million km2 have been designated as protected areas (PA). While initially the establishment of PAs in mountain areas was motivated by the desire to preserve mere wilderness and uniqueness, later the need was recognized to conserve biodiversity and sustain ecosystem services in particular freshwater supply (Tsering and Wahid 2011; Kollmair *et al*. 2005, MA 2005).

4.2 Extent

Worldwide, mountainous areas cover nearly 40 million km2, or about 27% of the Earth's land surface (UNEP-WCMC 2002) with Antarctica and Greenland accounting for about 7 million km2, based on the definition of mountains by Kapos *et al*. (2000) (UNEP-WCMC 2002; Kollmair *et al*. 2005). Around 20% of the world’s human population lives in mountains or at their edges, 90% of them are in developing countries or countries in transition. More importantly, despite of the general remoteness and hence relatively smaller population in mountain areas, about 90 million mountain people live in poverty and are vulnerable to food security (MA 2005).

Most mountains are located in the Northern Hemisphere and in temperate-sub-tropical latitudes. Much of the world’s greatest mountains are found in the enormous Eurasian landmass (UNEP-WCMC 2002). Eurasia also has the most extensive inhabited land area above 2,500 m elevation, in the Tibet (Xizang) Plateau and adjacent ranges. All of the world’s mountains above 7,000 m in height are in Asia, and all peaks above 8,000 m are situated in the Greater Himalaya range extending along the southern rim of the Tibet Plateau. Excluding Antarctica, South America has the second most extensive area of high elevation land, formed by the mountains and basins of the Central Andes. Antarctica and Greenland also constitute major mountain areas, due to their extent and thickness of their icecaps. Other important but relatively smaller mountain systems are located in Africa, Australia and New Zealand (MA 2005).

4.3 Hydrological processes in the mountains

The hydrological processes in the mountains are modulated by great topographical and climatic variability over short distances and distance from oceans (Whiteman 2000; Tse-ring *et al* 2010; Nogues-Bravo *et al*. 2007, 2008). Mountains intercept the global circulation of air and effect wind, precipitation and temperature patterns beyond their geographical boundaries into continental mainlands (Woodwell 2004). They modify air circulation and create their own winds by setting up regional and local pressure systems. Mountain and valley breezes interlock in diurnal circulation that can become strong enough to influence climate and temperature (Bothe *et al*. 2011). In general, mountains guide approaching air masses upward, and as temperature falls, the air is able to hold less water vapour, leading to increased precipitation on the windward side and a reduction on the lee side (the 'rain shadow' effect). About 24% of terrestrial precipitation falls in mountain regions (except Antarctica) (UNEP-WCMC 2002). Precipitation maxima vary and occur at different altitudes in different mountain regions of the world and are highest in humid tropical mountains.

At higher altitude levels precipitation occurs in the solid state and forms snow packs, which under favourable condition can transform into ice and form glaciers. Snow accumulates on the canopy in proportion to leaf area index, but most falls directly to the soil surface where it may accumulate to great depths. Fresh snow contains between 6 and 35% water by volume. The density of a snowpack increases to a maximum when snowmelt begins, at which time the water content per cubic meter is uniform and the temperature throughout the pack is isothermal at 0°C.

Mountain glaciers and icecaps comprise of about 684,294 km3 of frozen water (excluding the Antarctic and Greenland ice sheets) (Dyurgerov and Meier 2005; WGMS 2008) and are essentially natural freshwater reservoirs, gradually releasing water that has been stored as ice for many tens or hundreds of years. Melting of glaciers and snow is driven by the energy balance which in turn is determined by precipitation, air temperature, *albedo* (fraction of solar energy or shortwave radiation that is reflected from the Earth back into space), and radiation, and all with more or less strong dependence on altitude.

The orographic effect that causes increasing precipitation amounts in the mountain contributes to 40-60% of all freshwater supplies (FAO 2011; Bandyophayay *et al* 1997) through surface and sub-surface flow network or groundwater aquifers. The downstream water supply from mountains is influenced by its steepness of the terrain, geological and soil properties and vegetation and often, mountain water exerts a strong influence on the hydrological characteristics of downstream river basin, and the timing and volume of water generates distinct patterns of river flow, water temperature, suspended sediments, and hydrochemistry over annual, seasonal and diurnal time-scales (Milner *et al*. 2010; Hock *et al*. 2005) and impact downstream water use (e.g. hydropower generation, irrigation) specially in arid downstream areas or where water supply is controlled by special precipitation patterns like monsoon circulation. Hydrological contributions of mountains are also associated with the frequency and magnitude, and time-to-peak and duration, of floods in highly glacierized areas.
Box 1 shows the example of how the Himalayas are influencing the regional hydrological processes in South Asia.

BOX 1 – The Himalayas and South Asian hydrology
Land surface characteristics of the Himalayas play an important role in modulating the monsoon circulation and surface hydrology. The mountains shelter the Indian Subcontinent from the cold air mass of Central Asia and prevent frigid and dry arctic winds from blowing south into the subcontinent, keeping South Asia much warmer than other regions at corresponding latitudes around the globe. The Himalayas also exerts a major influence on monsoon and rainfall patterns (An *et al*. 2001) as they serve as a barrier for the moisture laden monsoon winds, preventing them from travelling northwards, thus facilitating timely and heavy precipitation in the southern part of the region (INCCA, 2010).

4.4 Hydrological functions of mountain ecosystem

Knowledge of the hydrological functions of mountain ecosystem is often fragmentary and incomplete. Extremely harsh terrain, difficult environment and large spatial and altitudinal variations in hydrological variables pose considerable difficulty in data collection for improving the understanding of the hydrological functions (Viviroli *et al* 2007) especially those related to the vertical height, seasonal variation in Leaf area index (LAI), and rooting depth of vegetation that affect water movement through ecosystems. This section attempts to identify the key hydrological functions of mountain ecosystems in terms of: surface water supply, soil water supply and flow regulation.

4.4.1 Surface water supply

The rich natural infrastructure of the mountains plays a significant role in surface water supply. The high-altitude cryosphere stores huge amounts of water as snow, ice and permafrost. These are unique reservoirs of fresh water which is released year round in perennial rivers. Almost all of the world's major rivers, and many of the minor ones, depend on water that starts the terrestrial phase of its cycle in mountain regions (Bandyopadhyay *et al*. 1997; MA 2005).

Based on detailed case studies, Viviroli *et al*. (2003) reported that globally mountain ecosystems contribution to mean annual river basin discharge varies between 32- 63% of the total. In some arid areas, mountains are estimated to supply as much as 95% of the total annual river discharge (Viviroli and Weingartner 2004). It is estimated that 23% of mountain ecosystem world-wide is essential for downstream region hydrology in the earth system context while another 30% have a supportive role (Viviroli *et al*. 2007).

The Hindu Kush Himalayan region, for example – known as the 'third pole' because of the concentration of highest masses of cryospheric components found outside the two Polar Regions – contains an estimated total ice cover of 60,000 km2 found in glaciers and an estimated 6,000 km3 of ice reserves which is equal to roughly three times the annual precipitation over the entire region. About 9.7% of the total glacier area – most in the valleys - is covered by debris which plays an insulating effect and reduces melting rates (Bajracharya and Shrestha 2011). Earlier, Dyurgerov and Meier (2005) reported ice areas of adjacent mountain ranges as: the Karakoram (16,600 km2), Tien Shan (15,417 km2), Kunlun Shan (12,260 km2), and Pamirs (12,200 km2). The Tibetan Plateau contains 36,800 glaciers, with a total glacial area of 49,873 km2 and a total glacial volume of 4,561 km3 (Yao *et al*. 2007). The glaciers that feed the Ganges, Brahmaputra, and Indus rivers in the dry season (Immerzeel *et al*. 2010) cover more than 32,000 km2. The contribution of snow and melt ice to Himalayan rivers is conservatively calculated to be between 500 and 515 km3 per year from the upper Himalayas alone. Snow and glacier melt comprises up to 50% of the annual flow in the Indus basin (Table 4.1).

Table 4.1: Contribution of glacier melts to the Himalayan river flows

Rivers	Basin Area (sq.km)	Annual mean discharge (m3/s)	% of glacier melt in river flow
Brahmaputra	651,335	21,261	~ 12
Ganges	1,016,124	12,037	~ 9
Indus	1,081,718	5,533	Up to 50
Mekong	805,604	9,001	~7

Salween	271,914	1,494	~9
Tarim	1,152,448	1,262	Up to 50
Yangtze	1,722,193	28,811	~18
Yellow	944,970	1,438	~2

Source: ICIMOD, 2011

In the European Alps, melting of snow and ice produces high, substantial discharge in summer, supported by low evaporation due to high elevation. Mean annual contribution from mountain area to total discharge varies from 26% in the Danube to 53% in the Po River (Kohler and Maselli 2009). Mountains also help to determine flow patterns and hydrological processes in many of the world's lake, river and wetland ecosystem, and maintain the sea level (e.g. Jacob *et al.* 2012; Meier *et al.* 2007). Increased melt can reduce glacier mass, providing short-term increases in melt-water contribution to downstream river flows. However, such increases will eventually decline, if available ice area reduces (Stahl and Moore 2006; Wanchang *et al*. 2000). The 1.4 million people of La Paz and El Alto (Bolivia) depend mostly on water supplies from surrounding glaciers located above 4,900 m above sea level, and 75% of the electric power for these cities is generated by the hydropower plants on the eastern escarpment of the Andes. Summer runoff variations in the downstream lowlands are moderated through highly regular melting processes and long-term compensatory water storage in the mountains.

Mountain forest exchange moisture with the atmosphere, which is important in controlling local and regional climate, especially precipitation thus modifying the water cycle. Plants delay snowmelt and infiltration into the soil through rooting, as well as through the associated soil fauna and decomposer communities. Natural mountain forests reduce peak runoff and local flooding to a certain extent. Natural forest cover, however, is significantly impacted by anthropogenic activities. While in the tropics natural forest cover was reduced by 6.8% between 1990 and 2000, temperate areas have experienced a 1.2% increase, mainly due to European mountain areas where the importance of protecting mountain forests in contributing to watershed protection, hazard prevention, tourism and other economic benefits has been recognized. (Kohler and Maselli 2009).

Montane cloud forests have particular relevance for the water cycle as they capture moisture from fog or clouds by which they add large amounts of water to the hydrological system which in turn sustains the forests. For example, in the Peruvian Andes a third of the endemic mammals, birds and frogs live in the cloud forests. Soil infiltration and bioremediation of water influence water quality. For example, forest buffers along agricultural lands can reduce nitrate concentrations in runoff from field by 5-30% per meter width of the forest.

High altitude mountain wetlands play an important role in in capturing and retaining melting snow or ice and, wherever possible, rainfall, releasing water progressively and therefore acting as suppliers and regulators of water for an entire basin (Trisal and Kumar 2008) and directly impact hydrological regimes and the wetland associated biodiversity. They maintain water quality, regulates water flow (floods and droughts). For example, the majority of China’s six million ha. mountain wetlands are peat lands which function like a sponge consisting of more than 90% water, and form major reservoirs of water maintaining water levels in streams, rivers and adjacent grasslands.

4.4.2 Soil water

Mountain ecosystem services are essential for soil water and related functions that maintain the hydrological balance in downstream areas. The spatial and temporal variation of soil infiltration capacity varies dynamically due to the spatial heterogeneity of the linkage between vegetation and soil moisture. Vegetation stabilizes the soil and affects the surface flow. If vegetation is removed, or changes its elevational extent, overland flow and erosion may occur and increase; this increases both stream flow and stream sediment concentration. Narrow and highly incised valley-bottoms often limit the extent of riparian zones, a key landscape position for nitrogen transformation (Cimo and McDonnell 1997). In addition their influence on surface water supply, mountain forests through litter, faunal activity and distribution of soil macropores often control the extent and magnitude of infiltration and subsurface flow. In the headwater catchment of alpine cold regions of Heihe River Basin in China, about 76% of the precipitation is transformed into groundwater or interflow and then concentrated into the river channel (Yong-gang Yang *et al*. 2012). The root systems and decomposer macro fauna of many tree species contribute to the increased infiltration of water into soils. Deep-rooted trees remove more soil water for transpiration, creating a larger soil water storage buffer, which may contribute to reducing peak runoff. Particularly in drier areas, trees redistribute water through their root systems vertically and horizontally to areas of lower soil moisture at night.

4.4.3 Flow regulation

Globally, mountain ecosystems regulate runoff generation and water movement from the cryosphere and are able to minimize year-to-year variability when catchment areas are moderately (10-40 %) glaciated. The limnological conditions and faunal distribution in the mountain areas impact lateral flow direction and soil moisture distribution, since gravity dominates total water potential in steep mountain terrain. Often, short flow response times to precipitation and snowmelt results in flashy surface runoff (overland flow) and subsurface flow that can generate floods. For instance, large-scale felling of trees in the mountain areas of the upper Brahmaputra basin have altered the riverine ecosystem drastically, as a result of which, the river has become heavily silted and floods are one of the most common natural disasters in the basin (Boruah and Biswas 2002).

Runoff regeneration from the mountain varies depending on the vegetation types. Runoff is generally lower from forested areas than from areas with less vegetation and, except on steep slopes with high sediment yield; erosion is often lower where natural forest occurs. For example, maximum surface runoff during heavy rain in the Austrian Alps is 40-80% lower in forests than pasture (Price *et al*. 2011). The timing of snowmelt is also a major determinant for initiating the vegetation cycle of many alpine plant species (Prock & Korner, 1996; Myneni *et al*., 1997; Keller & Korner, 2003).

4.5 Economic value of water-related mountain ecosystem services

Any value of water-related ecosystem services entails identification of the eventual use of water for drinking, irrigation, industry, recreation, hydropower generation etc. Many of these services do not only impact upon the overall quantity of available water but also quality (e.g. hydropower generation and recreation). The economic value of water-related mountain ecosystem services is considerable in terms of provision of water of adequate quantity and quality, throughout the year and adequate supplies of electricity throughout the year. Major changes in the cryosphere in terms of snowline shift, duration of snow cover, increase in cryogenic hazards such as ice and snow avalanches, glacier recession, formation and break-out of moraine-dammed lakes, warming of perennially frozen ground, and thawing of ground ice directly impact water resources and hydropower generation. The reduction in glacier volumes can have a strong impact on dry-season water flow in rivers fed largely by ice melt (Immerzeel *et al* 2010), which will very likely affect the provisioning of downstream water for drinking, hydropower (see Box 2), and irrigation.

BOX 2. Hydropower

Hydropower provides 19% of the world's total electricity supply. Mountains offer green cost-effective sources of hydropower all over the world and there are renewed zeal for harnessing hydropower in developing countries, particularly in India and China (Pandit 2009; Grumbine & Xu 2011). Several hundred hydropower projects are now proposed in the Himalayan region, which could lead to capacity additions of over 150,000MW in the next 20 years (International Rivers 2008). In the Andean mountains, the runoff collected from just around 10.5% of its mountainous watersheds (around 389,190 km2) is translated into a significant regional hydroelectric capacity of at least 20,000MW. However, development of hydropower is critically dependent on availability of river flow of adequate quantity and quality in terms of sedimentation. In the moist tropical environment, mountain forest ecosystem protects fragile slopes from soil leaching and erosion (Price *et al*. 2011). Destruction of mountain forests or at least change their structure may diminish their protective functions against floods, landslides, and rockfalls. For instance, the current trend in surface temperature increase in the Alps is likely to create favourable conditions for forest damaging organisms, such as bark

beetles that can destroy the Alpine forests. Dry summers may be responsible for frequent forest fires thus severely damaging forests. Extreme weather events, droughts, wildfire and the incidence of insect-borne diseases are all likely to increase, further threatening the habitats of mountain organisms. Such trends clearly have consequences for hydrological and protective functions of mountain forests and impact hydropower development around the globe.
However, large-scale hydropower development in the mountain can impact the region's natural infrastructure and alter the timing, flow, flood pulse, oxygen and sediment content of water, and threaten ecosystem health, particularly by disrupting environmental flows i.e. water requirements to sustain ecosystems. In addition, agricultural and forested land is lost; inhabitants of flooded areas are forced to move; and animals and plant species lose their habitats and fish cannot migrate naturally. Furthermore, uncoordinated promotion of small hydropower plants as alternative sources of energy in the mountains may be detrimental to the ecology. CIPRA (2011) reported large-scale ecological damage for relatively low energy gains in the European Alps where about 75% of water abstracted is used for hydropower. Thus, without proper environmental assessment and compliance with ecological standards, hydropower development in the mountains may pose a significant threat to biodiversity and forests, along with habitat loss and degradation.

Over 65 countries use more than 75% of their available fresh water for agriculture. These include countries with large populations such as India and China, which rely heavily on mountain discharge (Viviroli *et al*. 2003). For instance, according to a Green India States Trust (GIST) (Gundimeda *et al*. 2006) study, the per hectare ecological value of soil nutrient conservation, flood control, and water recharge in dense forest is of the order of INR 5,860 (about US$ 125) in Himachal Pradesh and about INR 6,255 (about US$ 134) in Uttarakhand. They estimated that the value of water related services rendered by five Indian Himalayan States (Jammu and Kashmir, Himachal Pradesh, Uttar Pradesh, Sikkim and Arunachal Pradesh) in 2003 was over US$ 1 Billion. The largest value accrued was due to flood benefit (US$ 493 Million) (see Table 4.2).

Table 4.2: Value of water-related services rendered by Indian Himalayan states, 2003 (million US$)

State	Nutrient loss	Water recharge	Flood benefits	Total
Arunachal Pradesh	278	68	324	670
Himachal Pradesh	47	11	54	112
Jammu and Kashmir	55	-	64	118
Sikkim	12	3	15	30
Uttar Pradesh	31	12	36	80
Total	423	95	493	1,010

On the other hand, water related hazards in mountain areas can have large-scale impacts associated with high costs. Glacial lake outburst floods (GLOFs), as occurring in the Himalayas, constitute a substantial risk to downstream communities and their economies and infrastructure such as hydroelectric power schemes. For example, rehabilitation of roads damaged by the outburst flood of Zhangzambo glacial lake in Tibet in 1981 cost US$ 3 million. The power supply was cut for 31 days, and the traffic was blocked for 36 days. Similarly, the outburst flood of Dig Cho glacial lake in Nepal in 1985 resulted in physical damage of property and infrastructure amounting to US$ 4 million (Khanal *et al*. 2009).

The global communities have realized that the mountains play an important role of providing water resources to the communities living in the mountains as well as downstream areas (Viviroli and Weingartner 2004, Pagiola *et al*. 2005a, b, Muñoz-Piña, *et al*. 2007, Pagiola 2008, Fisher *et al*. 2010, Arias *et al*. 2011). Now there is a growing interest on rationalizing the importance of mountains as a source of water and other ecosystem services across the globe (Wang *et al*. 2010). Latin America has the longest running and most robust experience in the realizing economic values of water-related services provided by the mountains. The prevalence of aligned programs in Latin America reflects several established conservation organizations' active role in seeking innovative financing for their projects (Stanton *et al*. 2010). In 2008, the economic value of watershed services in Latin America averaged US$100 per hectare per year.

4.6 Climate change-biodiversity-water cycle interaction

The mountain ecosystems of the world with their complex terrain and steep climatic gradients are likely to undergo highly heterogeneous responses to climate change. However, still little is known on the influences of concomitant changes in climate and biodiversity on a range of ecosystem goods and services such as the freshwater (Wolf *et al*. 2011). With much of the biodiversity located in the plains of the globe depleted in the last century, potential degradation of remaining biodiversity largely confined in the mountains will significantly impact on global ecosystems. Paleologic records indicate that climate warming in the past has caused vegetation zones to shift to higher elevations, resulting in large ecosystem changes in terms of species composition (UNEP/GRID-Arendal 2005) and introduced weeds. Simulated scenarios for temperate-climate mountain sites suggest that continued warming could have similar consequences. Species and ecosystems with limited climatic ranges could disappear in most mountain regions with changes in the extent and volume of glaciers and the extent of permafrost and seasonal snow cover. Lenoir *et al*. (2008) estimated that climate change has already resulted in an upward shift of average 29 meters per decade in optimum elevation of species. Figure 4.1 shows a comparison of current vegetation zones at a hypothetical dry temperate mountain site with simulated vegetation zones under a climate-warming scenario.

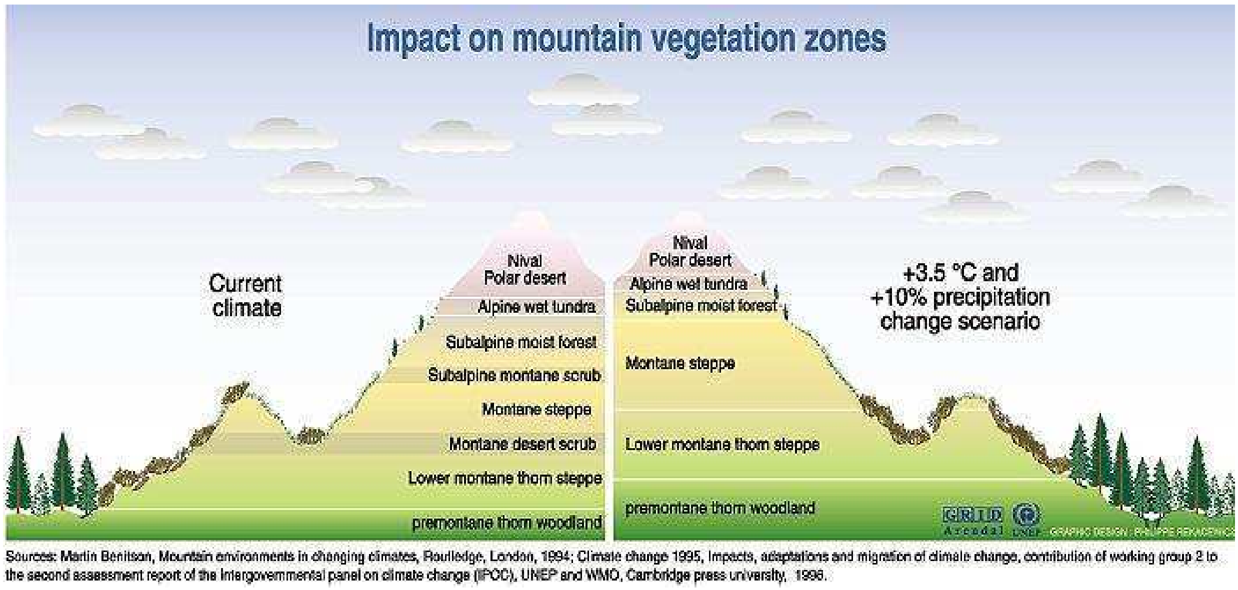


Figure 4.1: Climate change impact on vegetation zones at a hypothetical dry temperate mountain site (source: UNEP/GRID-Arendal 2005)

In the European Alps for each 1°C of temperature increase, the duration of snow cover is expected to decline by several weeks at mid-elevations which will induce major changes in the species composition (Bates *et .al*. 2008). As a result, species adapted to cold may become extinct through competition and habitat loss (CIPRA 2009). Furthermore, higher temperatures at higher altitudes could accelerate soil decomposition and change soil water infiltration pattern. In the Tibetan Plateau signs of degraded grassland have been attributed to warmer temperature, changes in combination of temperature and precipitation, decreasing glaciers, melting and overgrazing (Shang and Long 2007). Increasing temperatures and reduced rainfall also may have degraded large areas of peat lands dominated mountain wetlands which are susceptible to damage as a result of drainage or modification of the hydrological regime. In the future permafrost degradation will likely cause a drier ground surface (Cheng and Wu 2007). These changes will significantly affect soil properties and thus the water cycle (Wang *et al*. 2006).

4.7 Global and regional policies for sustaining mountain ecosystem services

The global community recognised the importance of mountains at the United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro, Brazil in 1992 through adoption of Chapter 13 in Agenda 21. Chapter 13 underscored the role of mountains in global sustainable development as well as the importance of ecosystems services provided by the global mountains. The chapter also expresses serious concerns related to the biodiversity degradation of many mountains. Similarly, the World Summit on Sustainable Development (WSSD) also dealt with mountain ecosystems and advocated in Paragraph 42 of the Plan of Implementation of the WSSD that: “Mountain ecosystems support particular livelihoods, and include significant watershed resources, biological diversity and unique flora and fauna. Many are particularly fragile and vulnerable to the adverse effects of climate change and need specific protection.”

The Conference of the Parties (COP) to the Convention on Biological Diversity (CBD) adopted the ‘Programme of Work on Mountain Biodiversity’ (PoWMB) as Decision VII/27 at its 7th Meeting, held in Kuala Lumpur in February 2004. The PoW on Mountain biodiversity was re-emphasised by CoP-10 in Nagoya, Japan in 2010. Since the advent of this Programme of Work, significant progress has been made in networking, mobilising, and influencing programmes related to mountain biodiversity (Chettri *et al* 2008, Sharma *et al*. 2010). These provisions have provided ample ground for addressing the pressing issues and challenges of ensuring mountain ecosystems services.

At the regional and local levels initiatives like the Alpine Conventions, Carpathian Convention, regional cooperation initiatives in the Himalayas (Sharma *et al*. 2010; Zomer *et al*. 2010), Bhutan Summit, Altai mountain (Badenkov 2011) and other advocacy initiatives are playing key role to bring mountain agendas in the policy arena for sustaining the water related ecosystem goods and services provided by the mountains.

However, widespread and continued ecosystems degradation in the mountain region imply that current policy regime and management framework on mountain biodiversity conservation must consider ecosystems, not just species. Water – or more generally ecosystem services, which is linked also to biodiversity – should also be included to face abrupt and disorganizing climate change. There is a need to think out-of-the-box and mainstream (align) the linkage between water and mountain biodiversity into global and regional political-economic initiatives. An integrated multidisciplinary approach at landscape level supported by an intensive capacity-building process involving conservationists, policy planners, decision makers, and local stakeholders is needed to ensure water availability to conserve the rich biodiversity of the mountains. Research and policies should be linked to livelihoods and local knowledge. Payment for ecosystem services is emerging as a tool to support mountain communities in sustaining the water related ecosystem services provided by the mountains. Best practices need to be identified and developed based on evaluation of current examples. Practitioners and policy makers should be engaged in funding research and in encouraging a development agenda that takes mountain communities and their livelihoods into consideration.

4.8 Management options

In recent years, significant progress has been made towards sustainable natural resources management in the mountains. On one hand, conservationists from across the globe are advocating for the cause of Mountains for Rio+20 for the inclusion of substantive support. And on the other hand, mountain people are strengthening the better understanding through various research frameworks such as Global Change and Mountain Regions (GLOCHAMORE), Global Observation Research Initiative in Alpine Environments (GLORIA), Global Mountain Biodiversity Assessment (GMBA) (Gurung 2006; GLORIA 2012; GMBA 2012 respectively). In 2010, more than 450 mountain experts have revisited the mountain issues based on which mountain research needs were identified such as increased focus on effective responses and innovations while maintaining the core competence in forecasting and observation; assessment of different drivers of change and their impacts on ecosystem services; improved understanding of human movements, sociocultural drivers affecting collective behaviour, and incentive systems; and in particular increased understanding of linkages between social and ecological systems. (Gurung *et al*. 2012).

The landscape or ecosystem approach, as advocated by the Convention on Biological Diversity (CBD) and reiterated through the Programme of Work on Mountain Biodiversity, is an important strategic framework provided by the CBD to address the conservation and management of mountain biodiversity. This approach, which has gained impetus (Chettri *et al*. 2009; Sharma *et al*. 2010; Worboys *et al*. 2010), requires increased regional cooperation, in part due to the biophysical nature and heterogeneity of mountainous regions, inter-linkages between biomes, habitats, and sectors, and the strong upstream - downstream linkages related to the provisioning of ecosystem services. Effective implementation of such regional cooperation has worked in countries of Western Europe under the Alpine Convention as well as in central and Eastern Europe under the Carpathian Convention, in both cases their focus is on conservation and protection linked to sustainable development. Similar efforts are underway in the Andes and in mountain areas in the Caucasus and Balkans. Also, in the greater Himalayas, efforts are made towards sustainable management of transboundary landscapes and their ecosystems as promoted by ICIMOD in the landscapes of Kanchenjunga, Brahmaputra-Salween, and Karakorum-Pamir which have been selected due to their relevance for biodiversity.

A worldwide proven and acknowledged approach to sustainably addressing the linkages between ecosystem components including vegetation, soil and water, and human systems is the integrated watershed management approach. A watershed is defined as the geographical area drained by a water course, ranging from micro-watersheds e.g. a farm crossed by creek up to large river or lake basins. As such, watersheds integrate conditions and processes over certain areas and determine the functionality of their ecosystems and the water yield for river systems, which provide essential freshwater for aquatic life, agriculture, hydropower generation, and industrial and domestic use for mountain and downstream populations (MA 2005). Hence, the sustainable management of a watershed is crucial to fulfil basic needs as well as to sustain and improve the livelihoods and economies of local and downstream communities. Incentive-based mechanisms such as Payments for Ecosystem Services (PES) can offer effective ways of compensating upstream communities for sustaining ecosystem services, in particular freshwater provisioning, to downstream communities provided that adequate institutional frameworks are in place. With increasing evidence and awareness about climate change impacts, watershed management needs to fulfil multiple objectives such as mitigating and adapting to climate risks, outlining a path to sustainable production of goods and services required by the communities, and sustaining the natural resource base. High mountain watersheds are of particular importance as larger communities in downstream areas depend on the various goods and services the high altitude areas provide. Effective watershed management requires taking into account the linkages between upstream and downstream areas while involving the local communities. Due to the high dependence of human well-being on well-functioning watersheds, watershed degradation constitutes a serious threat to sustainable development. Since watershed boundaries generally do not coincide with political boundaries, watersheds need to be managed at the transboundary level. In Tajikistan, for example, the Community Agriculture and Watershed Management Project aims at integrating local production improvement with global environmental objectives, protecting globally significant mountain ecosystems by mainstreaming sustainable land use and biodiversity conservation within farming systems and rural development programs.

Public schemes, international donors, and Research and Development institutions have increasingly invested in watershed management at different scales to ensure environmental stability, and socio-economic benefits from the relevant policy and practices. Through the World Bank support the mountain state of Himachal Pradesh in India is investing US$ 60 Million in watershed development. In its ‘Bhutan 2020’ policy document, the Bhutan government named watershed management as the “single most important strategy to maintain the resource base to support the national economy” (Jamtsho and Gyamtsho 2003). Particularly the participatory multi-stakeholder planning process was considered effective in the development of sustainable watershed management plans as it would help to adequately address the interdependence of issues related to natural resources management in the watersheds, ownership of resources, development efforts and conflict resolution. Emphasis needs to be given to considerations of environmental flows or water requirements by ecosystems which are a key element when it comes to managing biodiversity and water in an integrated way. In the mountains of Lesotho, the Mohale Dam has been designed in a way that it releases water of variable quantity and quality with the aim to provide occasional flooding downstream to provide essential environmental flows. In the face of climate change, there is an increasing need to develop further such integrated and holistic approaches for managing mountain ecosystems which sustain the flow of ecosystem services through integrating innovative solutions for climate change adaptation be it ecosystem-based such as sustaining highland wetland systems that provide various services including water regulation and habitat for biodiversity or new technologies such as drip irrigation systems.

Furthermore, in recent years, there has been increasing realization that protected areas and the biodiversity therein are critical sources of ecosystem goods and services and the values of these protected areas are important building blocks of economic development (TEEB 2010). The ecosystem valuation is necessary to bring sub-optimal decisions of the market agents to the notice of the policy makers so that we can put effective policy measures for biodiversity conservation based on market (dis)incentive approach.

4.9 Policy recommendations

- The overwhelming complexity of interactions between environmental change, water cycle and biodiversity in the mountain regions and social-environmental response calls for increased linking of expanded scientific research and sustainability policies. Future policy must anticipate these interactions by promoting long-term monitoring and data collection on climate and hydrology; and closing knowledge gaps by promoting substantiated studies of species composition, evolutionary and adaptive responses of species and/or biodiversity; and assessment of productivity and carbon dynamics in different ecosystems in mountain areas;

- To address changes in the water cycle as a consequence of climate change and the effects on biodiversity, climate change impacts should be incorporated as a critical factor into biodiversity conservation, and *vice-versa*. For example, plans for mitigating climate change through use of renewable energy systems (e.g. hydropower) should consider the potential effects of these systems on biodiversity;

- The improved understanding about biodiversity-water cycle interactions has important implications for how we manage and govern them. The current ‘paradigm’, in which water and biodiversity are managed separately, is obsolete. A comprehensive watershed perspective on extended landscapes and basins, considering upstream (mountain areas) and downstream (lower lying areas) linkage, should be promoted to integrate policies targeting biodiversity conservation and maintenance of natural water cycle;

- Global comparability in research data and findings as well as policy frameworks and regulations needs to be enhanced to understand the functioning of mountain biodiversity in the context of global climate change and basin-wide water resources management. Research on hydrological data is important for understanding how the water sources of the headwaters are connected to river systems, and for their sustained maintenance. For this, a workable definition of mountain areas will need to be agreed and adopted internationally;

- Research efforts need to increasingly focus on key linkages between upstream (mountain areas) and downstream (lower lying areas), their interdependencies, exchanges of goods and services and innovative policy and practice approaches to conventional issues (e.g. PES) to develop long-term sustainable solutions. Technological, financial, and institutional support should be increasingly directed to developing mountain countries through global mechanisms such as the

Global Environment Facility (GEF), the National Adaptation Programmes for Action (NAPA), and the Global Climate Change Alliance of the EU.

CHAPTER 5 Urban ecosystems

5.1 Urbanization: drivers and extent

Humans have cohabited in towns and cities for millennia (Carter, 1977). Urbanization and its reliance on the water cycle is not a 21st century phenomenon. The first urban revolution occurred thousands of years ago giving rise to the "great river civilisations" of the Tigris-Euphrates, the Nile, the Indus-Ganges and the Yellow River (Ito, 1997). The development and prosperity of these early urban centres depended on the agricultural potential of irrigated environments and the associated logistical advantages of anastomosing and braided channels (Oates, McMahon, Karsgaard, al-Quntar, & Ur, 2007). There is increasing evidence that the collapse of these early civilisations and the urban and agricultural systems which they supported, was initiated by hydrological changes and reductions in rainfall of up to 30% in a tract of the globe extending from Europe to the Indus (Cullen, *et al*., 2000; Weiss & Bradley, 2001). There is also evidence which suggests that in some cases desertification initiated by hydrometeorological changes may have been accelerated by changes in lad use and over grazing by livestock as migratory populations sought more favourable agricultural conditions (Weiss, *et al*., 1993). Ancient history demonstrates the precarious nature of the relationship among biodiversity, hydrology and urbanization.

Today, the urban population is growing at an unprecedented rate setting the social, political, cultural and environmental trends of the world (UN Habitat, 2011). These trends embrace both the good and the bad (Florida, 2005). Historically *Homo urbanus* was in a minority (Newman & Lonsdale, 1996). As recently as the 1950s, only three out of ten people resided in urban areas. However, just 50 years later, more than one-half of humanity now lives in towns and cities (Cohen, 2010). By 2050 the number of urban inhabitants is expected to reach 6 billion (United Nations, 2011)(Figure 5.1).

Estimates based on the interpretation of satellite imagery place the global extent of urban areas between 0.2 and 2.4% of the terrestrial land surface (Potere & Schneider, 2007). In the year 2000 cities with a population greater than 100,000 contained in excess of 600 million people and their total built up area (based on average densities of almost 3,000 persons per square kilometer) was over 200,000 square kilometers (Angel, *et al*., 2005). With the predicted increase in global urban population it is expected that land will be converted to urban areas at a rate of approximately 20,000 square kilometers per annum. It has been estimated that every new *Homo urbanus* results in the direct conversion of 500 square meters of non-urban land (Angel, *et al*., 2005). Whilst differences in the rates of urban land expansion vary from continent to continent and are driven by a variety of factors, on a continuum of human impacts urbanization represents the most irreversible of all land uses (Seto, *et al*., 2011).

On-going land conversion will continue to generate permanent and irremediable impacts on both biodiversity (Alberti, 2010; Hansen, *et al*., 2005) and the hydrological cycle (Fitzhugh & Richter, 2004; Hoekstra A. Y., 2009). Urbanization can drive the direct loss of natural ecosystems such as forests and grasslands, the drainage and conversion of wetlands (Millennium Ecosystem Assessment, 2005) and especially floodplain areas (Tockner & Stanford, 2002; Zedler & Leach, 1998). The densification and sprawl of built structures, if left unchecked, can generate impacts across a range of hydrological processes. Cities can be an unsustainable drain on water resources (Braga, 2001; Geldof, 1997; Fitzhugh & Richter, 2004), deplete groundwater (Konikow & Kendy, 2005) and have a long history of being polluters of aquatic ecosystems (Hynes, 1960; Niemczynowicz, 1999). The increase in impervious areas increases run off rates (Leopold, 1968; Scholz, 2006), alters latent and sensible heat fluxes (Offerle, *et al*., 2006) and reduces their resilience to climate change (Stone, *et al*., 2010). These changes to the hydrological cycle can be proximal and distal or of short or long duration (Hollis, 1990; Kingsford, 2000; Fitzhugh & Richter, 2004). Predictions suggest that without radical investment and commitment cities in some regions of the globe will fail to satisfy their own water demands and the future protection of freshwater ecosystems (McDonald, *et al*., 2011).

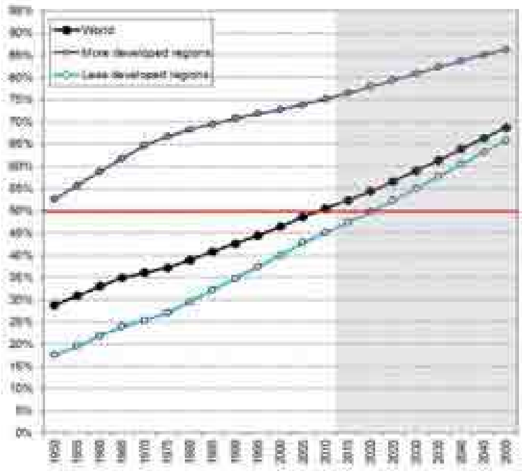


Figure 5.1. Percentage of Population Residing in Urban Areas (source**:** United Nations, 2011).

However, despite rapid expansion, cities can be hubs of national production and consumption driving the economic and social processes that generate wealth and opportunity (Van Vliet, 2002; UN Habitat, 2010). Compact, well-planned cities can generate resource efficiencies, reducing both energy consumption and carbon emissions (National Research Council, 2009). Increasingly measures are being introduced to increase water efficiencies (Rogers, de Silva, & Bhatia, 2002), to work with biodiversity to manage urban water issues (Brenneisen, 2006), to reduce their impact on the hydrological cycle (Woods-Ballard, *et al*., 2007), to mitigate and to adapt to climate change (McEvoy & Handley, 2006) and to produce water sensitive urban design solutions (Hoyer, *et al*., 2011).

5.2 Cities and hydrological processes

5.2.1 Water resources and footprints

Progressive urbanization can have a serious impact on the consumption of water resources as well as altering fundamental hydrological processes. In relative terms the direct water demands of cities are low, with domestic use taking about 10% and industrial use about 20% of all water withdrawals compared to 70% for agriculture (WWAP, 2009). Whilst the total global amount of fresh water available is adequate to satisfy the current population's need, its availability is not evenly distributed. Estimates demonstrate that over two-fifths of the world's population currently resides in river basins where the per capita water supply is at a level where disruptions to water supply are common and frequent occurrences (Fitzhugh & Richter, 2004).For over a decade water demands have routinely exceeded supply in over 80 countries (Gleick, 1993). Unless current consumptive patterns alter radically the percentage of humans living in basins facing water stress will continue to grow as demand outstrips supply (Revenga, *et al*., 2000).

Cities are dependent on water resources drawn from areas and ecosystems extending well beyond their municipal boundaries. The water footprint of a city can be defined as the extent of water use in relation to the consumption by people and is closely linked to the concept of virtual water (Hoekstra & Chapagain, 2007; Allan, 1998). Assessments of city's water footprints quantify the flows of virtual water leaving and entering the urban area (Hoekstra & Chapagain, 2007). In addition to basic potable water supplies and sanitation needs, urban populations in Europe and North American consume a considerable amount of virtual water embedded in imported food and products. According to one calculation, each person in North America and Europe (excluding former Soviet Union countries) consumes at least $3m^3$ per-capita-per-day of virtual water in imported food, compared to 1.4 m^3 per-capita-per-day in Asia and 1.1 m^3 per-capita-per-day in Africa (Zimmer & Renault, 2003).

Water is vital for the production of almost everything upon which cities depend. An average car tyre requires about $2m^3$ of water to manufacture; a ton of steel calls for $237m^3$; and an egg requires about $0.5m^3$. Even in temperate, humid regions where rainfall is frequent, the piped in water supply for domestic, industrial and other uses is as great as the direct precipitation input to many urban areas (Lerner, 1990). As 'water footprints' grow, individuals, companies and entire cities, will need to face the threat that there may soon not be sufficient water to me*et al*l demands (Hoekstra & Chapagain, 2007; Revenga, *et al*.,2000) or that to meet their demands will require unsustainable impacts across a range of other ecosystems.

5.2.2 Influence on hydrological processes

A consistent theme permeating the literature on urban sustainability and resilience is that rather than being viewed as a human construct, cities need to be considered as dynamic and complex ecosystems which often arrange themselves along a variety of gradients (Tjallingii, 1993; McDonnell, *et al*., 1997). As with other ecosystems, they are not uniform or static and will be subject to changes in their land surface, water and energy consumption or spatial configuration. Such changes will alter flows of energy and materials, including waste, water and biodiversity (Savard, *et al*.,2000). Therefore the impact of cities on the water cycle is not confined to water resources and footprints but also drives in situ influences on physico-chemical hydrological processes (Newson, 1994).

Urbanization is not a single or linear process resulting in a single outcome (Konrad & Booth, 2005). Cities are complex and heterogeneous ecosystems which include a variety of land uses, including *inter alia* woodland and forests (McDonnell, *et al*., 1997), important grasslands for invertebrates (Wood & Pullin, 2002), small-scale agriculture (Smit, Ratta, & Nasr, 1996), wetlands for managing urban runoff (Scholz, 2006) and a variety of novel ecosystems (Kowarik, 2011). Urban areas characteristically also include extensive areas of impervious surfaces. Therefore cities can alter a multiplicity of hydrological processes. This is not new and was recognised some forty years ago by the eminent hydrologist Luna Leopold who concluded that of all land-use changes affecting the hydrology of an area, urbanization is by far the most forceful (Leopold, 1968).

The same hydrological flow paths exist in cities as they do in other ecosystems (Figure 5.2). However, anthropogenic modification greatly alters their physical dynamics (Hall, 1984) and water quality and hence their ecology (Konrad & Booth, 2005). A dominant feature of many urban areas is the increased area of impervious surfaces, leading to a decrease in infiltration and an increase in surface runoff (Leopold, 1968; Newson, 1994). However, there are other subtle ways that biodiversity, even within the highly modified landscape of a city can beneficially contribute the hydrological response of cities.

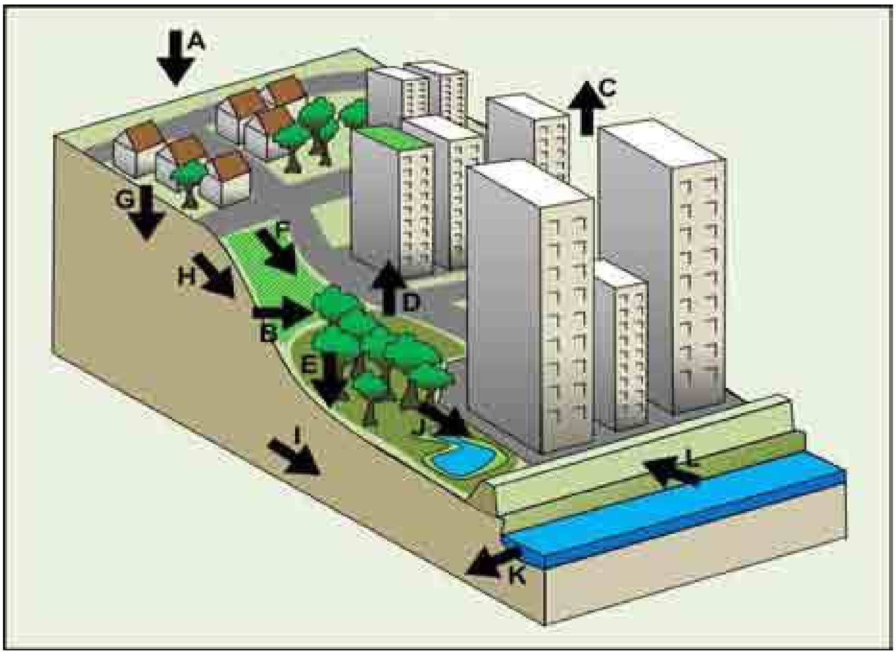


Figure 5.2. Flow paths in urban areas. (A=rainfall/snowfall; B=cloud-water interception; C=evaporation; D=transpiration; E=throughfall/stemflow; F=infiltration-excess overland flow; G=infiltration; H=lateral subsurface flow in soil strata; I=lateral subsurface flow in regoilith/rock; J=saturation overland flow; K=river/channel flow; L=overbank inundation).

Evaporation and transpiration

Cities possess built structures, such as buildings and roads, which combine to form micro-climatic features, and subsequently agglomerate with other buildings, gardens, car parks and sidewalks to create local-scale climatic regimes (Grimmond & Oke, 1999). These elements of the urban environment combine to modify significantly the fluxes of heat, moisture and momentum and further alter atmospheric processes through anthropogenic inputs of heat, water and pollutants (Taha, 1997).A common climatic result of urbanisation is the generation of 'urban heat islands' (UHI) where the air temperatures are relatively higher than the corresponding latitudinal and altitudinal rural values (Oke, 1988; Arnfield, 2003). Whilst the form and configuration of urban structures are known to influence significantly local weather, the issue is complicated because of the complexity of the urban terrain and the associated turbulence and energy transfer processes (Souch & Grimmond, 2006).

Traditionally, the role of evapotranspiration (*ET*) has been neglected in urban hydrological studies, primarily because of the dominance of built and impervious surfaces and the relatively reduced rates when compared with other natural or rural ecosystems (Grimmond & Oke, 1999). However, evapotranspiration, along with urban albedos which have been shown to decrease summer temperatures by up to 4^{o}C (Taha, 1997), is a potential moderator of urban micro-climates (Solecki, *et al*., 2005) and particularly as mitigation for the negative health impacts caused by urban heat islands (Haines, *et al*.,2006).

Increasingly attention is being directed towards the 'oases' effect of vegetated areas within the urban environment where as well as providing direct shading and other benefits, under the right conditions, evapotranspiration from urban green space can reduce air temperatures by 2-8^{o}C in comparison to surrounding areas (Taha, 1997; McPherson, 1994). Under certain circumstance the latent heat flux (λE) can be sufficient that the sensible heat flux (H) becomes negative, causing the air above vegetated surfaces and over drier built environments to supply sensible heat to the vegetated areas (Taha, 1997; Rizwan, Dennis, & Liu, 2008). This experience has been recorded for suburban lawns (Suckling, 1980), urban woodlands and parks (McPherson, 1994) and green roofs (Oberndorfer, *et al*., 2007). In some of these situations the Bowen ratio (ratio of H to λE) can tend towards the negative resulting in a significant reduction in air temperatures. The increased evapotranspiration rates associated with vegetated urban spaces has been shown to produce a maximum cooling of 1.6^{o}C from urban parks in Hong Kong (Tong, *et al*.,2005) and 2^{o}C from urban grasslands in Tokyo (Ca, *et al*.,1998) . A simulation study of ten cities in the United States of America demonstrated the relative significance of additional tree planting in metropolitan areas as a method to reduce ambient air temperature through elevating evapotranspiration rates (Table 5.1).

Table 5.1. Number of additional trees planted in each metropolitan area and their simulated effects in reducing the ambient temperature. (Source: Taha, *et al*., 1996)

Location	**Millions of additional trees in the simulation**	**Millions of additional trees in the metropolitan area**	**Maximum air temperature reduction in the hottest simulation cell (^{o}C)**
Atlanta	3.0	1.5	1.7
Chicago	12.0	5.0	1.4
Los Angeles	11.0	5.0	3.0
Fort Worth	5.6	2.8	1.6
Houston	5.7	2.7	1.4
Miami	3.3	1.3	1.0
New York City	20.0	4.0	2.0
Philadelphia	18.0	3.8	1.8
Phoenix	2.8	1.4	1.4
Washington, DC	11.0	3.0	1.9

Despite the predominance of impervious surfaces and built structures, evapotranspiration represents an important flux within urban environments, acting as an energy sink, and hence reducing urban temperatures. Not only do urban green spaces provide a range of ecological and social benefits within a largely artificial environment they can be important drivers of climatic functioning assisting to modify the local hydrological cycle (McPherson, *et al*., 1997).

Interception and infiltration

The high degree of impermeable surfaces associated with urban buildings, roads and other structures, including unpaved compacted soils, reduces infiltration to ground and subsequently rates of groundwater recharge (Gregory, *et al*.,2006). For instance Lindh (Lindh, 1983) stated that “infiltration to ground is markedly reduced” [in urban areas]. Lerner (1990) suggested that whilst infiltration and recharge are heavily modified in the urban environment, they are influenced by two distinct pathways: the altered natural pathway and the water supply-sewage pathway, which can result in recharge higher than pre-urbanization rates.

Urban water supply and sewage-waste water systems are often a complex of interconnecting systems. Water can be piped to cities from distant watersheds, boreholes bring groundwater into the network, water supply pipes leak, stormwater is discharged to ground via soakaways, infiltration basins and permeable pavements, septic tanks discharge waste water to ground and over irrigation of urban parks and gardens contributes to increased evapotranspiration and infiltration rates (Figure 2). These various interconnected pathways can carry large flows and account for high percentages of the urban water cycle and the balance of urbanized aquifers (Table 2). For instance in Doha it has been estimated that urban aquifer received over 87% of its recharge from park irrigation, leaking mains and discharges from sewers and septic tanks (Lerner, 1990) (Table 5.2).

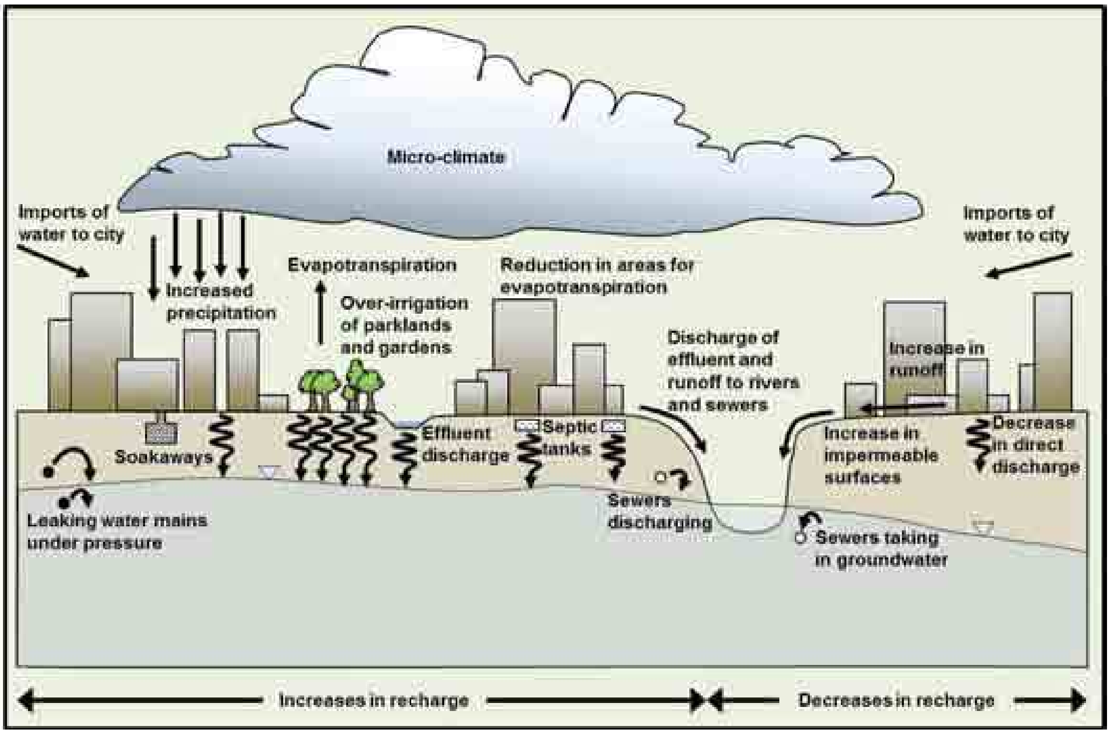


Figure 5.2. Urban effects on groundwater recharge. (Redrawn from Lerner, 1990).

Biodiversity has a key role play at the interface between the natural (but usually highly modified) and anthropogenic urban hydrological pathways. Even the biological action of acidogenic, acetogenic and methanogenic bacteria which drive the anaerobic digestion process in septic tanks assist in improving urban waste water quality prior to discharge to ground. Vegetated infiltration basins, grass swales, buffer strips, as well as rain gardens and green roofs, all influence interception and infiltration rates, ultimately assisting in moderating stormwater runoff in urban areas. For instance, trees can intercept and store rainfall on their leaves and branches and root growth and decomposition can increase the capacity and infiltration rate of urban soils (Xiao, McPherson, Simpson, & Ustin, 1998). In Santa Monica, California, for example, municipal forests intercepted 14.8% of a winter storm event and 79.5% during a summer storm and across the city's streets and parks trees intercepted 1.6% of the annual precipitation (Xiao & McPherson, 2002).

Vegetated infiltration basins and swales are now routinely applied as elements within urban sustainable drainage systems (Woods-Ballard, *et al*., 2007). The use of a variety of plant species in such design features, including trees, can assist in improving infiltration to ground (Gregory, *et al*.,2006), reducing soil compaction (Bartens, *et al*.,2008), removing potentially polluting contaminants (Ellis, 2000) and attenuate the stormwater hydrograph (Scholz, 2006).

Often these features are used in combination with other elements such as permeable pavements and wetland systems (Scholz, 2006) in order to achieve water sensitive urban designs (Lloyd, *et al*.,2002). Such approaches to improving interception and increasing interception, for the tripartite roles of reducing flood risk, increasing groundwater recharge and improving water quality, have biodiversity at their core even within the highly modified urban landscape.

Table 5.2. Example of water balances of urbanised aquifers ($10^3 m^3 d^{-1}$). (Source: Lerner, 1990).

City	Lima	Doha	Bermuda
Area (km^2)	400	294	6.3
Recharge from:			

Precipitation	0	11.5	4.83
Rivers	280	0	0
Agricultural irrigation	-	0	0
Park irrigation	390	37.6	0
Leaking mains	340	25.3	0
Sewers and septic tanks	0?	17.6	3.13
Soakaways	0	-	-

Stream flow and overbank inundation
Many urban streams have been straightened, culverted or constrained between concrete banks so that effectively they are little more than open, artificial drains (Newson, 1994). In the United States of America estimates suggest that more than 130.000 km of rivers and streams are directly impaired by urbanization (US Environmental Protection Agency (USEPA), 2000). Widespread engineering work across urban centres, often to enhance flood conveyance and disaster risk reduction, whilst increasing the 'flashiness' of the hydrograph has reduced or removed connectivity of flows, sediment movements and the dispersal of species between the river channel and the floodplain (Gurnell, *et al*., 2007). Such actions result in predictable changes in the flow dynamics, water quality and ecological functioning of urban water courses, often manifest in consistent declines in the species richness of invertebrates, amphibians and fish communities (Paul & Meyer, 2001).

The physical alteration of urban streams often disconnects the biotic elements both along the longitudinal channel continuum (Vannote, *et al*.,1980) and through important riparian zones out to the wider floodplain (Gregory, *et al*.,1991). The loss or degradation of the linkage between the physical hydrology and the ecological components not only alters the contribution of stream flow and overbank inundation to the wider hydrological cycle it potentially undermines biological diversity at the landscape scale (Petts & Amoros, 1996).

Water quality
Urban development is a major cause of water quality degradation (US Environmental Protection Agency (USEPA), 2000; Paul & Meyer, 2001). Runoff from urban areas results in an increase in nutrient loads, metals, pesticides, organic contaminants and other pollutants (Gurnell, *et al*.,2007; Niemczynowicz, 1999). Additionally thermal regimes can be impacted negatively (Leblanc, *et al*.,1997) and levels of dissolved oxygen deficits can lead to fish kills (Gafny, *et al*.,2000). Often these impacts persist for some distance downstream of urban centres. A study on the Yarra River, Australia, indicated that the impact on urbanization on macroinvertebrate assemblage composition in riverbed sediments was more persistent downstream of urban centres than the positive effects of riparian forest cover, suggesting that restoration and management activities should focus on mitigating the negative impacts of catchment urbanization (Walsh, *et al*.,2007).

Green infrastructure, such as infiltration basins, sustainable drainage features and wetlands can be used within urban environments to mitigate the impact of polluted storm and waste water (Scholz, 2006; Woods-Ballard, *et al*.,2007). However in-stream water quality can remain poor if elements are not joined up and a holistic approach to managing water in urban environments fails to be pursued (Lloyd, *et al*., 2002; Gurnell, *et al*., 2007).

5.3 Urbanization, hydrological functions and ecosystem services

5.3.1 Urban ecosystem services

Whilst cities depend on wider ecosystems for the flow of energy, materials and water, they can also benefit from the ecosystem services generated from within municipal boundaries. Any urban area can be considered as a complex ecosystem where it represents a single entity in a state of flux (Tjallingii, 1993), or as a mosaic of individual ecosystems such as lakes, parks and gardens (Rebele, 1994). Often the habitats present in urban landscapes are novel or emerging and bear little resemblance to pristine natural habitats (Kowarik, 2011). The relationships among urban form, density, and the distribution and type of green space can influence both the delivery of ecosystem services and the nature of the beneficiaries (Tratalos, *et al*., 2007; Alberti, 2010).

Biodiversity conservation has traditionally relied on a system of protected areas (Heller & Zavaleta, 2009) in order to protect threatened and endangered species. The emphasis of this process has been outside of urban areas. Such a system is fundamental as a means to an end, where that end is the protection and maintenance of all forms of ever-decreasing wild biodiversity (Locke & Dearden, 2005). Whilst the maintenance and appropriate management of rural protected areas is essential for slowing down the global loss of biodiversity (Bruner, *et al*., 2001) biodiversity also exists outside of rural protected areas and inhabits urban spaces. Cities are not irrelevant to biodiversity conservation with many species living and commuting through urban areas (Kinzig, *et al*., 2005) and many protected areas lying within or contiguous to urban centres. However, many urban habitats, such as brownfield sites or roadside verges, which do not support protected species fail to get recognised within the classical protected area model yet these areas can still support a rich diversity of species and deliver important ecosystem services (Knapp, *et al*., 2008; Robinson & Lundholm, 2012). The majority of urban biodiversity conservation strategies aim at preserving and reconnecting remnants of native habitats and restoring native species. Whilst such approaches are essential, the question arises as to whether traditional approaches need to be supplemented as they fail to embrace the full range of urban nature (Kowarik, 2011).

Similarly, recognition of the importance of the common, local and non-iconic species, including the bacteria which drive many biogeochemical processes, can often remain subservient to exotic and appealing species (Ballouard, *et al*., 2011).This is in part an element of a wider perception issue where focus on flagship species can detract conservation attention away from the overall importance of non-charismatic biodiversity in broader ecosystem function (Clucas, *et al*., 2008) or the delivery of ecosystem services within urban landscapes (Bolund & Hunhammer, 1999).

Groundwater recharge

Despite the preponderance of impervious surfaces, precipitation falling on urban green spaces can directly recharge groundwater. Excessive irrigation of parks and gardens and leakages from water supply and sewerage networks can also provide indirect groundwater recharge (Lerner, 1990). A study conducted in Austin, Texas, demonstrated that direct recharge decreased from 53 $mmyr^{-1}$ prior to urbanization to 31 $mmyr^{-1}$ in post urbanization conditions. However, when indirect sources were taken into account an additional 85 mma^{-1} resulted generating a recharge rate in excess of urban development (Garcia-Fresca, 2006).

However, due to the complexities of urban water management systems, quantifying groundwater recharge is difficult. Often point sources are unknown or difficult to locate and measurements can be impossible in all but a few cases. Modeling and computation of mixing ratios based on hydrochemical signals has been employed to overcome issues of source location. In Barcelona, Spain, analysis of water samples from city aquifers suggest that rain falling on urban green spaces accounts for 48% of all recharge, including non-urbanized areas (17%), infiltration from runoff (20%) and recharge from the Besòs River (11%). The remaining 52% is composed of recharge from water supply network losses (22%) and sewerage network losses (30%) (Vazquez-Sune, *et al*., 2010).

Flood regulation

The conservation, restoration and creation of ecosystems both within and beyond urban areas can assist in the management and reduction of flood risk. In Japan it has been recognized since the early 2000s that non-structural interventions and catchment management options must be employed to combat the risk of flooding to highly urbanized areas (Takeuchi, 2002). A classic study from the Charles River in the United States of America demonstrates how the management of ecosystems within a broader watershed can deliver significant benefits to downstream urban areas (Faber, 1996).

The catchment of the Charles River is one of the most densely populated river basins in North America. Urban and suburban development from Boston, Cambridge and surrounding communities has destroyed much of the lower river's wetlands and natural landscapes. This has resulted in a reduction of natural water storage and significant downstream flooding in 1938, 1955 and 1968 causing millions of dollars' worth of damage. The United States Army Corps of Engineers commenced an analysis of the situation in the mid-1960s and discovered that wetlands still played a major role in storing excess floodwaters and reducing the potential for damage on the upper and middle portions of the Charles River (Doyle, 1986). However, despite an understanding of their value, wetlands in Massachusetts continued to be degraded and lost at a rate of up to 1 percent per annum. The destruction of wetlands in the upper River Charles basin not only extended flooding problems throughout the catchment and it exacerbated flooding in the lower basin and urban areas in particularly, as floodwaters, liberated from the buffering by wetlands, could move downstream more quickly (Faber, 1996).

In 1972, the Corps of Engineers commenced work to alleviate flooding in the lower basin by replacing the existing dam at the mouth of the river. A new dam and associated pumping station, which could divert high flows to Boston Harbour, was completed in 1978. The Corps' initial proposal for the basin also recommended the construction of levees and a second dam along the middle portion of the Charles River at an estimated cost of $100 million at 1970s prices. However, the 1968 flood had taught the Corps important lessons regarding the capacity of the wetlands to store flood waters. Based on an understanding of the capacity of wetlands to attenuate flooding, in 1977 the Corps began purchasing land and acquiring easements, prioritizing parcels by location, storage capacity and threat of development. By 1983, the Corps had purchased approximately 1,300 hectares and acquired easements on 1,975 hectares of private land. The protected area now includes over 75 percent of all existing wetlands in the Charles River watershed. In addition to the wildlife, recreational and economic benefits which have resulted from the protection of wetlands, estimates have suggested that the capitalized flood control value of wetlands within the Charles River basin was approximately $5,000 per wetland hectare at 1981 prices (Thibodeau & Ostro, 1981).

However, increasingly it is accepted that there is a role for urban green spaces to play in reducing flood risk within urban areas. For instance, in Dublin, Ireland, the use of natural green infrastructure to mitigate stormwater runoff has become mandatory in all new developments (O'Sullivan, *et al.*, 2012). Whilst quantifying the impacts of natural systems within the urban water cycle remains a challenge due to spatial complexities (Burian & Pomeroy, 2010), in the US it has been demonstrated that urban greening projects not only influence hydrological processes by reducing flood risk but also deliver commensurate benefits such as raising property values, invigorating local economies, boosting tourism, preserving farmland and safeguarding wider environmental quality (Lerner & Poole, 1999).

Many examples of integrated urban flood management are emerging which integrate biodiversity within sustainable solutions. In Dordrecht, the Netherlands, attempts to mitigate urban flood risk have been integrated into new and upgraded developments. Individual flood resilience measures have been adopted at the street and building level, often utilizing green technologies and natural infrastructure, which also deliver wider societal benefits, such as high amenity value and visual attractiveness. The work in Dordrecht has shown that these green measures may be directly more economically efficient than structural responses to flood management and indirectly provide additional economic benefits through multiple use values (Zevenbergen, *et al.*, 2008). In Philadelphia, United States, green infrastructure has been applied to address stormwater control. The added value of working with natural systems as compared to using a sewer tunnel across 50% of the city's impervious surfaces has been estimated at some US$2.8 billion over a lifetime of 40 years (Table 5.3) (Centre for Neighborhood Technology, 2010).

Table 5.3. Multi-value additional benefits associated with stormwater management green infrastructure (Centre for Neighborhood Technology, 2010).

Stormwater detention
Reduced energy for heating or cooling
Reduced health impacts from extreme heat events
Air quality improvements
CO_2 reductions (avoided and sequestered)
Urban heat island mitigation
Reduced energy use, air pollution and greenhouse gas emissions
Reduced ground conductivity (urban heat island and use of road salting in winter)
Reducing air pollution
Reduced noise pollution
Reduced potable water use
Increasing available water supply
Stormwater retention and pollutant removal
Increased property values
Recreation value
Avoided conventional infrastructure costs
Reduced wastewater treatment costs
Reduced flood risk damage
Increased groundwater recharge
Societal benefits such as crime reduction and improved exercise
Noise reduction
Public education opportunities
Biodiversity and habitat
Longer roof life

One aspect of biodiversity management which is gaining increasing prominence in urban developments is the integration of habitats within green roofs. Given that roofs account for 40 to 50% of impermeable surfaces in urban areas the potential to manage these areas for both flood risk mitigation and wider environmental benefits is clearly evident. Empirical studies have demonstrated that vegetated green roofs not only reduce the amount of stormwater generation they also suppress the peak flood hydrograph and extend it considerably beyond the duration of actual rain events (VanWoert, *et al*., 2005). A study in Brussels, Belgium, demonstrated a similar outcome suggesting that extensive roof greening on just 10% of buildings would yield a 2.7% reduction in peak runoff generation for the region (Mentens, *et al*., 2006). Increasingly green roofs are being considered as complex ecological systems which provide multiple benefits from stormwater management to climate cooling (Oberndorfer, *et al*., 2007), however, the evidence is also suggesting that well designed roofs can also contribute to the conservation of rare and threatened species (Kadas, 2006).

Water quality regulation
There is a large body of evidence demonstrating the value of green infrastructure within urban environments in improving water quality and mitigating pollution risk (Scholz, 2006; Ellis, 2000; Zedler & Leach, 1998). Often the use of natural systems can be cost effective in terms of both capital and maintenance costs, even without factoring the broader environmental benefits delivered. The implementation of an integrated constructed wetland to deal with domestic waste water from a housing development at Glaslough, County Monaghan, Ireland, not only effectively removed pollutants (Table 5.4) but it also provided three times the treatment capacity at a half of the capital costs as well as providing a range of amenity and recreational benefits (Doody, *et al*., 2009).

Table 5.4. Performance of an integrated constructed wetland at Glaslough, Co. Monaghan, Ireland (mean values February 2008 to August 2009) (Doody, *et al*., 2009).

	Influent (mg/l)	**Effluent** (mg/l)	**Removal efficiency** (%)
Biochemical oxygen demand (BOD)	837	5	99
Chemical oxygen demand (COD)	1179	37	97
Suspended solids (SS)	2544	9	99
Total nitrogen (N)	43.36	1.69	96
Total phosphorus (P)	7.91	0.31	96
Ammonium	34.04	0.34	99
Nitrate	9.81	0.19	98
Molybdate reactive phosphorus (MRP)	4.28	0.04	99

In the city of Fuzhou has faced many of the waste water problems associated with rapidly developing cities. A canal running through the city received heavily polluted waste waters and was characterized by unpleasant odours and floating solid waste. A decision was taken by the city government to pursue a low cost-low maintenance solution in favour of a traditional electro-mechanical engineered solution. A series of floating vegetation boxes, each planted with native flora, were installed along a 500m reach of the canal providing water treatment through the biophysically diverse surfaces provided by the plant root zones and biogeochemical interactions in the fabric media within the boxes. Waste water entering one end of the canal is recycled to the top of the canal for treatment and a low-intensity aeration circulates water and forces it passed the biologically active zones which are inoculated with beneficial bacteria. The system is now reducing suspended solids, improving water quality and providing an enhanced habitat and quality of life for local residents (Gaddis, 2003).

That Luang Marsh lies on the outskirts of Vientiane, Lao PDR, and provides a vast economic benefit to the city through the ecosystem services it provides (Gerrard, 2004). Not least of these services are the protection of the city from flood risk and the cleaning of waste water. Recent work by NGOs and the city government has aimed to improve water quality through the targeted restoration and construction of areas of wetland within the wider Marsh. Pilot study locations have been selected on four criteria: (1) to address existing problem areas related to waste water around That Luang Marsh; (2) to be in areas which will test constructed wetlands in a range of situations; (3) to be in areas where there is clear interest and ownership by local stakeholders; and (4) to have a pre-identified management scenario for after the wetland is constructed. Locations were selected for the pilot studies including treating waste from a local school, households, a pulp and paper mill and a brewery. Whilst some issues arose during the establishment phase, primarily associated with undertaking construction during the dry season resulting in leakage and managing seasonal extreme water level fluctuations, the sites have demonstrated effective water treatment as well as providing the local communities with wider benefits including education, aesthetics and biodiversity (Gerrard, 2010a; Gerrard, 2010b).

Other services

Green spaces and urban ecosystems contribute widely to the quality of human life within cities. Often the benefits of urban ecosystem services go unnoticed and are taken for granted by the uninitiated citizen and improvements in air quality, ambient temperatures and flood risk are not recognized in terms of the importance of biodiversity (Rodriguez, *et al*., 2006). Despite the growing body of evidence regarding their importance, the city-region environment presents many challenges for the structural integration of ecosystem services into landscape planning, management and design (de Groot, *et al*., 2010).

A study from Stockholm, Sweden investigated six ecosystem services performed by different urban habitats (Bolund & Hunhammer, 1999) (Table 5.5). Seven different different urban ecosystems were identified: street trees; lawns/parks; urban forests; cultivated land; wetlands; lakes/sea; and streams. Of these all of them regulated the urban micro-climate and provided recreational and cultural values. Wetlands were the only ecosystem to deliver all six of the ecosystem services under investigation.

Table 5.5. Urban ecosystems generating local and direct services relevant to Stockholm, Sweden. (Source: Bolund & Hunhammer, 1999).

	Street tree	Lawns / parks	Urban forest	Cultivated land	Wetland	Stream	Lakes / sea
Air filtering	X	X	X	X	X		
Micro-climate regulation	X	X	X	X	X	X	X
Noise reduction	X	X	X	X	X		
Rainwater drainage		X	X	X	X		
Sewage treatment					X		
Recreation / cultural values	X	X	X	X	X	X	X

The importance of the urban ecosystems in providing micro-climate regulation reflects the conclusions of the discussion on the importance of evapotranspiration in cities for reducing the urban heat island effect. The results from New Jersey, United States of America, demonstrate that urban tree planting can provide energy cost savings through the reduced need for electro-mechanical air conditioning (Solecki, *et al*., 2005). In certain areas where mature trees were present the cost savings were in excess of US$700 yr^{-1} for every hectare of woodland in the city. The cost savings also translate into avoided carbon emissions which were greater than 60 tonnes $ha^{-1}yr^{-1}$. In addition to the energy and carbon savings associated with climate regulation by urban trees, a study in Chicago, United States of America, estimated that in 1991, the city's trees removed an estimated 5575 tonnes of airborne pollutants which equated to value for air cleansing worth US$9.2 million (McPherson, *et al*., 1997). A similar study conducted in Capital Park, Sacramento, CA, estimated the annual dollar and energy savings for avoidance of space heating/cooling (as a result of evapotranspiration) avoidance of sewage treatment capacity due to reduced discharges resulting from storage within natural infrastructure, avoidance of stationary source air pollution control systems and avoidance of fertilization and soil catchment basins. The annual environmental benefits delivered by green infrastructure, and urban forests in particular, were estimated to range from US$10,000 to US$137,300 for Capital Park, which equated to approximately US$30 to US$389 per individual tree (McPherson, 1992).

Urban green space can also provide important social functions and cultural ecosystem services. “Boundary parks” which separate distinct urban communities have been seen to facilitate more opportunities for diverse cultural groups, especially among children and young adults, helping to reduce segregation and diffuse social tensions (Gobster, 1998). Just as boundary parks can assist with social cohesion and do not distinguish between social class, colour or creed, studies from Sheffield, UK, demonstrates that economically deprived groups and the older were the most likely to benefit from recreational ecosystem services (Barbosa, *et al*., 2007). A review of urban green infrastructure has demonstrated that issues of social cohesion, as experienced in Sheffield, often occur in parallel with a range of positive impacts on the hydrological cycle (Table 5.6, Centre for Neighborhood Technology, 2010) further emphasizing the integrated nature of social and ecological systems (Ostrom, 2009).

	Benefit																	
	Reduces stormwater runoff											Improves community livability						
Practice	Reduces water treatment needs	Improves water quality	Reduces grey infrastructure needs	Reduces flooding	Increases available water supply	Increases groundwater recharge	Reduces salt usage	Reduces energy usage	Improves air quality	Reduces atmospheric CO2	Reduces urban heat island effects	Improves aesthetics	Increases recreational opportunity	Reduces noise pollution	Improves community cohesion	Urban agriculture	Improves habitats	Cultivates public education opportunities
Green roofs	Yes	Yes	Yes	Yes	No	No	No	Yes	Yes	Yes	Yes	Yes	Maybe	Yes	Maybe	Maybe	Yes	Yes
Tree planting	Yes	Yes	Yes	Yes	No	Maybe	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Maybe	Yes	Yes
Bio-retention	Yes	Yes	Yes	Yes	Maybe	Maybe	No	No	Yes	Yes	Yes	Yes	Yes	Maybe	Maybe	No	Yes	Yes

Yes Maybe No

Table 5.6. Benefits delivered by different green infrastructure practices within cities (modified from Centre for Neighborhood Technology, 2010).

5.4 Cities and carbon

5.4.1 Carbon storage and sequestration in cities

Despite having limited spatial extent and distribution in an urban landscape, green spaces can still be important stores and regulators of carbon. A study of the importance of above-ground carbon storage in the city of Leicester, UK, demonstrated that over 97% of all carbon was stored in trees (Davies, *et al*., 2011). However, 40% of the tree-cover comprised four common native species (*Crataegus monogyna* Jacq. (14%), *Fraxinus excelsior* L. (12%), *Acer campestre* L. (7%) and *Prunus avium* L. (7%)) which would be expected to be associated with various urban habitats. Whilst some of this tree cover occurs in urban parkland and associated areas which are afforded a degree of protection, this study highlights the fact that the benefits provided by common species often remain inadequately accounted for within urban areas. Based on the Leicester study, the contribution of cities, through the important ecosystem service of above-ground carbon storage in common tree species, to national carbon storage estimates was undervalued by an order of magnitude (Davies, *et al*., 2011).

Alongside the hydrological benefits, similar carbon storage benefits have been calculated for urban forests in Canberra, Australia. When Canberra was selected as the site of the new capital for Australia in 1911 the area was a relatively treeless grazed plain. Successive tree planting has established large areas of urban forest. In the mid-1990s the city managers commissioned the development of a computer based systems to collect, store and interpret data on trees planted in public areas (Banks, *et al*., 1999). A detailed investigation was undertaken to ascertain the carbon sequestered in trees during the Kyoto commitment period (2008-2012).

The computer modeling calculated that in the period 2008-2012 street trees would sequester 130,000 tonnes of carbon and Park trees a further 172,000 tonnes (Brack, 2002). Based on a nominal value of

US$10 per tonne it has been possible to allow a comparison between carbon sequestration and the value of urban forests for energy reduction. The predicted energy savings of street verge and parkland trees was estimated at US$1.57m, with a further benefit of US$0.32m in carbon sequestration. When other benefits such as hydrological and pollution amelioration were factored into the equation the value of urban trees in Canberra over the period 2008 to 2012 was estimated at US$20.05 million (Brack, 2002).

5.5. Policies and Practices for Urban Ecosystems

5.5.1. Rio+20 Summit Declaration on 'The Future We Want'

In June, 2012, 20 years on from the Rio earth Summit in 1992, the Rio+20 Summit was held in Brazil. It was attended by representatives from 191 countries and concluded with an announcement of declaration titled 'The Future We Want'. This declaration considered green economy in the context of sustainable development and poverty eradication as one of the important tools available for achieving sustainable development. The declaration also recognized the key role that ecosystems play in maintaining water quantity and quality and recommended supporting actions within respective national boundaries to protect and sustainably manage these ecosystems. Additional the declaration recognised that, through the conservation of natural and cultural heritage, well planned and developed cities can promote economically, socially and environmentally sustainable societies.

To deliver on the commitments enshrined in the Rio+20 outcome document requires the development and implementation of key policy methods and best practices for the management and sustainable use of low-carbon resource efficient urban ecosystems that can contribute to the green economy by minimizing environmental degradation and ecologic resource depletion and at the same time enhancing human welfare and social equality.

5.5.2. Policy Instruments and Practices for Urban Ecosystems

Despite the substantial science-base, renewed efforts are needed to reverse the impact of urbanization on hydrological processes and functions and ecosystem services, and to ensure that the efficient and sustainable use of natural resources is based on holistic rather than reductionist approaches (Munda, 2006). A range of policy instruments exist to facilitate this challenge (Turner & Daily, 2008). The policy instruments available for urban ecosystem conservation, and sustainable and efficient use can be categorized as regulatory (i.e. command-and-control) approaches, economic instruments, information and capacity building, as shown in Table 5.7.

Regulatory Approaches

The objective of the “Strategic plan for Biodiversity 2011-2020 and the Aichi Targets” is to address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society. Target 2 states that: “By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.” Within cities, these objectives are commonly integrated within regulatory approaches for urban ecosystem protection and management. A cornerstone of urban ecosystem policies and strategies is the development of planning tools and the enforcement of regulations. Some of the regulatory approaches are reviewed below.

Table 5.7. Policy instruments for urban ecosystem conservation, and sustainable and efficient use.

Regulatory (Command-and-Control) Approaches	Economic Approaches	Information Sharing Approaches	Capacity Building Approaches
• Low impact development / SUDS • No net loss policy • Design standards for green infrastructure • Vulnerability assessment • Spatial planning (e.g. ecological corridor) Planning tools and regulations (e.g. Indices for city sustainability; • Establishment of protected areas(PAs);	• Biodiversity offset / bio- banking • Urban CDM • Payments and subsidies for ecosystem services • Carbon banking system • Soil Security • Taxes and fines to curb undesirable behaviors	• Urban knowledge sharing platform • Corporate Environmental accounting	• Green governance • Public-private partnership (PPP) • Training program

Low Impact Development (LID) is a widely adapted regulatory instrument for urban ecosystem management (Dietz, 2007). It includes elements such as bioretention, permeable paving and vegetated swales. LID utilizes innovative stormwater management principles that are modeled after nature through the management of rainfall at source using uniformly distributed decentralized micro-scale controls (U.S. EPA, 2000b). The elements within LID also reflect approaches to Sustainable Urban Drainage Systems (SUDS) developed in the United Kingdom (Scholz, 2006) and Water-Sensitive Urban Design (WSUD), a term used in Australia (Wong, 2006).

An advantage of LID techniques is that they emphasize the control or at least minimization of changes to the local hydrologic cycle or regime. Regulators can use LID to address a wide range of wet weather flow issues, including combined sewer overflows (CSOs), national pollutant discharge elimination system permits, total maximum daily load (TMDL) permits, nonpoint source program goals, and other water quality standards. Local Permitting agencies can use LID as a model in revising local zoning and subdivision regulations in favor of more cost-effective, ecologically sound development practices. Developers can achieve greater project success and cost savings through the intelligent use of LID, and designers can apply these techniques for innovative, educational, and more aesthetically pleasing sites. LID's long-term success often has much more to do with the knowledge, skills, and creativity or the site designers than what the property owner does or does not do (Dietz, 2007).

Design standards for green infrastructure have been widely adopted as a regulatory instrument for urban ecosystem development and management (American Planning Association, 2006). For instance, in mid-2001, the British Town & Country Planning Association (TCPA) published "A Programme for Sustainable Communities" calling for the positive planning and delivery of a great number of homes to higher standards in sustainable social cities – or 'sustainable communities'. This demand enhanced levels of "biodiversity, renewable energy and energy efficiency", it set out a vision, which "…above all, sees our communities as integrated with the natural environment rather than set against it" (TCPA, 2004). The aim of the guide is to provide guidance on how to maximize the opportunities for biodiversity in the planning and design of sustainable communities. It takes the user through the design process, presenting a toolkit of best practice that can be tailored according to the scale of the development opportunity. The guide for sustainable communities reviews UK case studies, which have been paired with international examples, and provides useful lessons for the improved management of biodiversity and hydrological cycles within the context of green infrastructure.

A cornerstone of most national and local ecosystem management policies and strategies is the *establishment of protected areas* (PAs) (Hansen & DeFries, 2007). Protected areas can also be an integral component of local green infrastructure. Designing networks of PAs connected by ecological corridors is also important for restoring, maintaining or enhancing ecological coherence, the natural adaptive capacity of ecosystems and their role in the hydrological cycle. This is particularly important in urban areas where PAs are often small and vulnerable. In establishing and managing PAs, local governments should ensure that the legal and customary rights of indigenous people, local communities, and other stakeholders are fully respected, and consider the important role local and indigenous communities can play in the management of PAs and as a source of local and traditional knowledge (OECD, 2012).

Economic Approaches
Biodiversity offsets and payments for ecosystem services (PES) are two of the more commonly used economic instruments for managing biodiversity and ecosystem services. Innovative carbon financing approach includes urban CDM and carbon banking system.

Biodiversity offsetting or bio-banking is increasingly being used as an economic instrument for urban ecosystem management. Offsets are conservation activities designed to deliver measurable biodiversity benefits to compensate for residual losses caused by project development, after appropriate prevention and mitigation measures have been undertaken. Biodiversity offsets can operate in either a regulatory or voluntary framework (Gardner, et al., 2012). Australia, Brazil, Canada, China, France, Mexico and South Africa, among others, have developed guidelines or incorporated biodiversity offsets into their legal framework, while several industry leaders have voluntary incorporated offset policies into their corporate strategy. These include Rio Tinto, BHP Billiton, Anglo Platinum, and Shell (OECD, 2012).

Some governments have introduced incentive mechanisms to encourage or require mitigation and compensation for adverse impacts. Whilst the long-term implication of market-driven policies remains uncertain (Robertson, 2006), in some cases, new markets for ecosystem services or biodiversity ‘credits’ have been established, in which the private sector may be both significant buyers and sellers, due to their role as land managers as well as their responsibility for land disturbance. Wetland mitigation banking in the United States was one of the first such systems to be established; it has accumulated considerable experience and has been refined over time. Under this scheme, developers are obliged to compensate for damage to wetlands, either directly or by purchasing credits from third parties, based on the restoration of wetlands in the state watershed. Although the approach is still evolving, the market for US wetland credit is currently estimated to be worth between US$ 1.1 and 1.8 billion annually. Although the approach is still evolving, the market for US wetland credit is currently estimated to be worth between US$ 1.1 and 1.8 billion annually (TEEB, 2010).

Several Australian states have introduced similar schemes, whereby disturbance of native vegetation and impacts on species habitats may be compensated by an appropriate offset, generated by active conservation or restoration projects. Examples include the Biobanking scheme introduced in New South Wales in 2008; and the Bushbroker Scheme in Victoria, which has so far facilitated more than AU$ 4 million in trades.

Given that cities are responsible for 60 to 80% of global energy usage, *urban clean development mechanisms (CDM)* are a vital yet still evolving innovative approach to achieving sustainability and stemming environmental degradation. There is the potential for CDM to provide additional funding and investment sources which could facilitate the participation of cities in international carbon markets (UNEP, 2012). Urban CDM is a mechanism which gives financial incentives and provides Certified Emission Reductions (CERs) as much as amount of emission reduction at the city-scale compared with baseline emissions or business-as-usual at the city level (Kim, 2012). The CERs can then be traded and sold and used by industrialized countries to meet their emissions reductions targets and defined under the Kyoto Protocol.

A general lack of financial incentives for cities to reduce greenhouse gas (GHG) emissions has been identified by several organizations (UNEP, 2012). So far, existing carbon finance mechanisms such as CDM do not target local authorities specifically. For example, only a small number of examples of CDM projects that could be labeled “Urban CDM” (i.e. CDM projects that have a city-wide approach in one or several sectors with multi-planning dimensions) have been identified so far. Such a carbon financing mechanism at local level could provide a key incentive for city authorities to manage urban ecosystems in a carbon smart manner among cities in particular in developing countries. The water-biodiversity-carbon nexus within cities could provide an essential element of future urban CDM schemes.

Payments for Ecosystem Services (PES) are a rapidly emerging instrument used to reduce the loss or enhance the provision of ecosystem services. They are defined as “a voluntary, conditional agreement between at least one ‘seller’ and one ‘buyer’ over a well-defined environmental service - or a land use presumed to produce that service” (Wunder, 2005).

For instance, downstream hydroelectric utilities that use clean water as an input to production pay upstream forest managers to ensure a sustainable flow of this service. Numerous examples are appearing including the Tasmanian Forest Conservation Fund programme in Australia, and the Sumberjaya Watershed programme in Indonesia.In case of water system in Han river in the capital area of the Republic of Korea, burden charges are paid by residents living around the downstream in order to compensate for financial disadvantages caused by the restriction of land in upstream regions (World Water Assessment Programme, 2009).

PES can be potentially much more cost-effective than indirect payments or other regulatory approaches used for environmental objectives. Increasingly the relationship between cities and the surrounding rural landscape is providing opportunities for the development of rural employment and income through the sustainable supply of goods and services for cities through PES schemes (Gutman, 2007).

Carbon Banking Systems treat sequestered carbon in the same way that banks and financial institutions consider capital whereby carbon ‘deposited’ in ecosystems is exchanged for an annual payment and those that need carbon offsets can borrow from the carbon bank. Carbon banking is one of the representative environmental policies of Gwangju Metropolitan City in the Republic of Korea, and is being considered widely as a component in low-carbon green cities. The example from Gwangju Metropolitan City is the first case of a climate response model among Korean local governments for which a local government, a local bank, and citizens work together (Kim, 2012).

The carbon banking system targets the GHG emissions in the household sector. In Gwangju’s case, 47% of CO_2 emissions are generated by households and the commercial sectors. This system is therefore introduced as governance in order to both reduce GHG emissions in the household sector and raise awareness amongst citizens. The amount of CO_2 reduced by each household is calculated by a local bank, and then granted back to homes as carbon points that can be used as cash. As of July 2012, 270,000 households have joined this activity since its initiation in 2008. In 2010, 145,000 households have succeeded in saving energy thus received 2.694 million points which is equivalent to a reduction in GHG emissions of 20,550 tons.

In order to take root of this system Gwangju city is focusing on the civil education. ‘Carbon Coordinators’ visit households for a consultation on the energy use and promote carbon banking system. 'Green Home Designers' are also trained to give citizens directions how to reduce GHG emission at home.

Information Sharing Approaches
Numerous fora and platforms exist at a range of scales to enhance the promotion of *Knowledge Sharing Approaches*. While international agreements such as the Convention on International Trade in Endangered Species and the Convention on Biological Diversity are negotiated at the global level, the input of cities is highly important in ensuring implementation at the local, national, and regional levels.

The Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES) is the interface between the scientific community and policy makers which has the knowledge generation catalyst function for biodiversity and ecosystem services. In a meeting of scientific organizations interested in IPBES convened by ICSU and hosted by UNESCO, participants recommended that the word "Knowledge" be used throughout, rather than "scientific information", since knowledge is a more inclusive notion, including scientific knowledge as well as other forms of knowledge such as local, traditional, and indigenous knowledge (ICSU-USGS, 2011). As the recognized global mechanism for pooling knowledge on biodiversity and ecosystem services, IPBES has an important role to play in ensuring urban issues and relationships are understood and reported.

Initiated by the World Bank, the Urbanization Knowledge Partnership (UrbanKnowledge.org) is a global knowledge partnership that provides local authorities with a platform to put the world's best knowledge and data in the hands of policy makers and practitioners, in order to harness urban growth for better development outcomes. Their four thematic pillars include economic: rural-to-urban transition, social: social inclusion and mobility, environmental: sustainable urban growth, and governance: creating accountable cities and towns.

Cities can also take advantage of the wealth of information on biological biodiversity collected by information networks such as the Global Biodiversity Information Facility (GBIF). The GBIF was established in 2001 to encourage free and open access to biodiversity data through the internet. The GBIF suggests that by the end of 2016, it should have demonstrated an unquestionable rationale for 'Biodiversity Information Facilities' becoming a permanent infrastructure in every country and region. Both science and society stand to benefit enormously from a fully operation 'Global Biodiversity Information' accessible to all (Global Biodiversity Information Facility, 2011).

Capacity Building Approaches
Numerous organizations, both intergovernmental and non-governmental, provide technical support on biodiversity. UNEP's World Conservation Monitoring Centre(UNEP-WCMC) can provide expertise, tools, techniques and information, and works to establish networks to promote conservation and information exchange. The IPBES will also provide an interface between the scientific community and policy makers. One of the four main functions of the IPBES will be to prioritize key capacity-building needs to improve the science-policy interface.

5.6 Future management opportunities

5.6.1 The reality check

There are, undoubtedly, practical reasons why a great many things cannot be done within cities. But values are shifting in the face of impending needs; and needs are a powerful agent for change. Increasing consciousness of biodiversity, water and a changing climate will undoubtedly modify views on long-term versus short term urban ecosystem management (Alberti, 2010).

There are needs and opportunities for improved management of ecosystems or natural infrastructure to support sustainable water security for cities including:

i. Institutional and governance needs;

ii. Policy, regulation, and legal issues;
iii. Cities and urban authorities as a source of financing and other incentives for ecosystem conservation and/or restoration;
iv. Cities as generators of ideas and solutions.

Institutional and governance needs
In many societies, the engineering works in cities have been undertaken for and at the command of a ruling elite (Douglas, 1983). Even in democratically governed cities, the engineering agencies can become powerful, dominant lobbies in local communities, able to argue the case for more expenditure on engineering works, such as larger water supply reservoirs, bigger urban free-ways, or more elaborate power generating and supply systems, including nuclear power stations. Thus, while cities look to engineers to provide the essential urban services and to monitor and control pollution, some engineering agencies have probably pushed engineering solutions, such as structural flood control measures and river channelization, to the exclusion of other alternatives for natural infrastructure.

There are needs for considerations over who wins and who loses from the effects of basic urban service provision or environmental improvement. Multiple layers of governments and a wide range of public and private stakeholder experts should be included to build buy-in and crucial partnerships for coordinated biodiversity strategies. The private sector in these interactions should be taken into account.

Policy, regulation and legal issues
Policy, regulation and legislation affect the location, design and management of natural infrastructure in cities. Thus, there are needs for mandatory approaches for natural infrastructure provision and access to similar funding sources (Benedict & McMahon, 2002).

Cities and urban authorities as a source of financing and other incentives for ecosystem conservation and/or restoration
Cities and urban authorities can play an important role to help stimulate the emergence of biodiversity markets and the adoption of market mechanisms for biodiversity conservation as a source of financing and other incentives for ecosystem conservation and/or restoration (Grimm, et al., 2008). Effective financial response to biodiversity loss and ecological degradation for critical natural infrastructure involves subsidies to ecosystem services which can increase economic efficiency and reduce the fiscal pressure confronting governments.

Cities as generators of ideas and solutions
Cities are commonly a nation's generator of innovation and ideas (Marceau, 2008). Cities also lend themselves to the development and implementation of pioneering work by experts and civil society. Areas that can be addressed by experts in the future include regular updates on biodiversity change projections, improved mapping and geographical data, and periodic assessments of biodiversity impacts to inform a broad spectrum of urban ecosystem policies and programmes. To create practical solutions for these ideas, innovative urban biodiversity strategies can be prepared. Pioneering work undertaken in South Korea represents one of the initiatives resulting greater awareness of biodiversity and ecosystem services and the need for accurate scientific information. The pilot work in Korea led to an experimental UN-Habitat Sustainable Cities Programme for creating eco-cities from 1999 to 2001 including "Biodiversity: The Hanam Strategy" (Kim, 2000).

Using Hanam City as a pilot study, the strategy covered both urban area and agricultural areas and had the following four extensive objectives which are being mainstreamed in cities across the world:

1) Identify, protect and improve existing sites that are the targets of biodiversity interests;
2) Encourage the creation of new areas for biodiversity conservation;
3) Promote the sustainable use of biodiversity;
4) Allow every class of a local community to participate in biodiversity conservation and sustainable use.

These sentiments were captured and expanded in the Shanghai Declaration on Better Cities, Better Life promulgated in 2010 which stated that “tackling the challenges of urban development, innovation offers solutions and the concept of ‘Cities of Harmony’ embodies our dreams”. One of the key tenets of the Shanghai Declaration was that the role of cities as generators of technical and scientific innovation should be promoted as a path to urban development. The pursuit of such as approach sees the role of biodiversity and the urban ecological environment as an asset that needs to be integrated into urban planning and administration in order to accelerate the delivery of sustainable development.

5.7 Policy recommendations

Even in highly modified urban environments, biodiversity can still play a crucial role in modifying the hydrological cycle. Common, local and non-iconic species and novel ecosystems can be important providers of ecosystem services, and the regulation of the hydrological cycle in cities.

Urban planners and managers need to promote ecosystem services and resilience to deliver multiple benefits. Recommendations arising from the urban ecosystem work include a broad range of policy-relevant suggestions, some focused on critical infrastructure and some focused on broader-scale actions.

i. Mainstream and integrate promotion of ecosystem services and resilience into land-use and urban planning to enhance synergies and prevent trade-offs. For example, energy saving spatial planning provides greater benefits to biodiversity and ecosystem services than others. Biodiversity loss and ecosystem degradation have other impacts, including on climate change, water quantity and quality, and human health. Understanding these inter-linkages and interactions can help policy makers identify potential policy synergies and trade-offs, and thus enable more co-ordinated and strategic decision making. (OECD, 2012)
ii. Incorporate hazard mitigation in the overall planning process to reduce environmental hazards in the city. The embedding of hazard mitigation into the planning process is going to vary from country to country, depending on political structures and social and cultural traditions. (Douglas, 1983).
iii. Assets impacts on the urban system in monetary terms with a common yard-stick, cash. The economic value of ecosystem services provides a better basis for linking ecosystems to growth objectives and will be used by the local government to direct policy in the future.
iv. Conduct a review of standard and codes to evaluate their revision to meet hydrological challenges and biodiversity loss, or the development of new codes and regulations that increase the city’s resilience to climate change. Develop design standards, specifications, and regulations that take biodiversity and ecosystem service change into account, and hence are prospective in nature rather than retrospective.
v. Establish a biodiversity and ecosystem monitoring program to track and analyze key pressure factors, state, impact and response (PSIR), and evolving-knowledge indicators in cities, as well as to study relevant advances in research on related topics. This involves creating a city-wide network of monitoring systems and organizations and indicator database for analysis.

CHAPTER 6 Managing Natural Infrastructure for Enhanced Water Security in Agriculture

6.1 Introduction

The general circulation of the moisture-laiden atmosphere is drive by temperature and pressure gradients, powered by solar energy, and modified by the land-sea-topographic configurations on the earth's surface. The amount of precipitation and its seasonal distribution, along with the temperature regime at any location on the earth's surface, are dictated by the earth's tilted annual orbit around the sun and by the daily rotation on its own axis. Consequently, there are uneven distributions of resources of weather and climate, soil and terrain, water, and biodiversity across the whole globe. These resulting biophysical resources, and their local and global interactions in space and time, constitute the global natural infrastructure that has evolved over geological time. The natural infrastructure comprises ecosystems or biomes (MEA, 2005) that are life-sustaining and offer ecosystem services to humankind who depend on them for their survival and well-being. Human societies utilise the resources and services offered by ecosystems within the global natural infrastructure to meet human needs, including for food, water, energy, and for socio-cultural, economic and environmental service development. In the process the natural infrastructure is altered and managed to meet desired goals, as has been happening ever since humankind has been living on the earth. Given the fact that what passes for 'natural' has already had some man-made interventions and modifications, these changes are accelerating with anthropogenic climate change. The Millennium Ecosystem Assessment (MEA, 2005) showed that some two-thirds of our ecosystems are already moderately to severely degraded, and the challenge for the future is to find ways to manage them optimally and sustainably to meet human needs, including their rehabilitation or further purposeful evolution where necessary and possible.

Water security for sustaining food and nutrition security is important at all levels of human organization, from households to villages and nations to international and global levels. Additionally, water security for maintaining agricultural production of industrial raw materials for economic purposes needs to be taken into account when assessing overall national water security. Thus water security concerns more than just food. There are various significant and competing non-agricultural water uses that figure into any assessment of water security as non-agricultural water uses are needed to generate employment and income.

Over the past 50 years, agriculture has met the rapidly rising demand for food and raw material for industries, but this has come at a high cost to the environment and quality of life. For example, agriculture which is the major user of water is also its single largest polluter of water supplies, in addition to contributing to soil and atmospheric pollution Along with growing water scarcity due to over-use and increasing competition, greater uncertainty stemming from climate change, and decreasing per capita availability of water due to rising population and increasing demand, there is widespread concern in the international community of the need to ensure a more sustainable use and management of water resources and the natural infrastructure that is a the source of water. However, there is not enough serious action in terms of agricultural land use at the national and international level to match this concern for water resources, both quantity and quality.

Land is an integral part of the larger natural infrastructure within which the water cycle is embedded. Globally, the water cycle and the nested regional, sub-regional and local water cycles within it are 'powered' by solar and gravitational energy. The solar-powered components of the water cycle involve the evaporation of water from the sea, from inland water bodies and other wetlands, and from land and vegetation surfaces. The evaporated water vapour eventually returns to the earth's surface from the atmosphere, due to cooling and condensation, in the form of precipitation, and this feeds into the landscape-level hydrological processes occurring within watersheds. The gravity-powered components of the water cycle involve infiltration, runoff, water retention in the soil, deeper drainage, and groundwater and stream flow (Figure 6.1). Both solar- and gravity-powered components of the water cycle are affected by the way that land is used and managed and by the physical, hydrological, biological and chemical properties of soil systems and their physiographic position in the natural infrastructure such as the watershed.

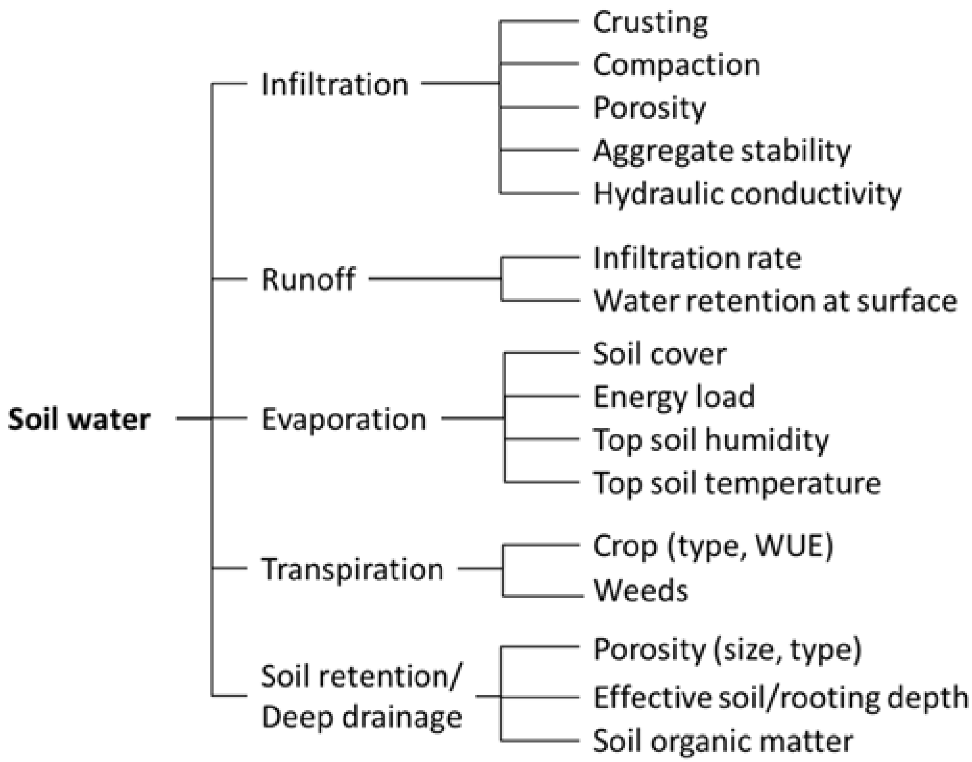


Figure 6.1 Processes and parameters affecting soil water

Agro-ecosystems are part of the natural infrastructure that humankind has developed by altering certain components of the natural ecosystems such as vegetation cover, soil health and productive capacity, water availability in the case of irrigated production, for the purposes of agricultural production of food and industrial raw material. Over the past two to three millennia, where-ever humankind has altered the natural infrastructure for agriculture, degradation of the natural resource base has ensued, and has invariably led to a loss in the productive capacity of the land, often leading to desertification and abandonment (Montgomery, 2007). The extent and form of agriculture that can be practiced across the world depend on the distribution of water availability and productive soils. However, the type of agriculture or other forms of land and soil management practiced has a direct and significant impact on the short- and longer-term water balances and on the real-time water regimes of farming, forest and pastoral locations as well as physiographic areas such watersheds and whole river basins, some transcending national boundaries.

In general, the natural infrastructure globally is declining in quantity and quality due to human activity, and instead of investing in enhancing it or at least ensuring that it is not depleted or depreciated, the opposite seems to be the case. In the case of agriculture land use, integrating water with soil, land and vegetation (including crops) or land cover management involves both the water cycle and the soil system. Solar energy and gravity are two forces, one exogenous and the other endogenous, that power the two physical 'legs' of the natural infrastructure. There is a third, biophysical 'leg' -- the microorganisms and the other biodiversity in the soil' – that affect/mediate how the land/soil and water resources interact and function – some of the energy that serves this third leg is simply recycling of biomass, with decomposition and uptake of the elements/nutrients: but management such as soil cover and aeration, or watering where water balance is in deficit, will make this a more effective physical/biological (i.e., biophysical) system, one which provides environmental services for agriculture and non-agricultural production and human needs.

Subterranean aquifers as part of the natural infrastructure serve as water storage facilities for humankind everywhere, and the groundwater they hold comes from: (i) precipitation water that has infiltrated into the soil and reaching the aquifers through deep drainage beyond the vegetation root zone, and from (ii) lateral gravitational flow of groundwater. The inflow or recharge of the aquifers depends on the characteristics of the soil and the geology. These underground water resources play a key role in water security in terms of quantity and quality including for domestic water security especially around large cities (IUCN, 2011) where water quality must be of acceptable standard for domestic use.

Mountainous regions also serve as water storage facilities in the form of 'water towers' in the form of glaciers and snow, and surface and groundwater, for a large proportion of the earth's population, particularly for people located in the downstream arid and semi-arid regions. Populations that live in the riparian and floodplain areas and rely on the mountainous regions for their water supply must ensure that upstream uses of natural resources including water do not reduce the water security, both in quantity and quality, of the populations living downstream and in the floodplains. With respect to the above water cycling, the sealing of land areas, including agricultural land with a reduced capacity of deep infiltration and recharge of the aquifers, human intervention can also disrupts the water cycles. But with correct management, this can be re-established and managed landscapes can come again close to natural environments, with a feed-back loop to water cycles. Mountainous regions capture moisture and, by forcing moist air to ascend to higher cooler altitudes, create precipitation. Yet, if the mountainous areas themselves are degrading, the precipitation cannot be maintained and is running off the mountains, resulting in flash floods downstream. The same situation appears in floodplains: if they cannot absorb the waters and store them underground, they become flooded without a longer-term contribution of the water to lifecycles. Besides the creation of precipitation, caused by mountains and also by forest areas due to the high transpiration rates, it is very important for 'water generating' landscapes to have a capacity to capture and store the water in the ecosystem, preferably underground.

Water resource management requirements for meeting water security needs at the national and regional levels depend on an array of domestic and civil, agricultural, economic, industrial and environmental activities and demands. This makes water resources and water resource management to meet multi-purpose objectives a complex challenge influenced by climate, vegetation cover, land use, and the competing demands for water which in turn depend on the social, economic and environmental factors.

Approximately 62% of the water falling on land is either stored within or evaporated from the soil and plants (Hartmann, 1994). Of the water stored on land, 25% is stored in the vadose and saturated zones of the soil, while a share of 74% is still retained in glaciers (Bockheim and Gennadiyev, 2010). These figures clearly indicate the importance of soil water for the Earth's land–water–energy balance and a possible positive feedback as the climate warms in the future (Koster *et al*., 2004; Huntington, 2006).

Water cycling is essential for all living organisms and for the planet, and is considered as a supporting ecosystem service (MEA, 2005). Water security implies an organised ability and capacity of the society to meet its desired needs for water and those of nature. Fresh water is therefore an essential provisioning ecosystem service from land, of which the society makes multiple, and often competing, uses including for domestic, agricultural, industry, energy generation, and environmental purposes. Agriculture currently accounts for 70% of all liquid water withdrawn from aquifers, streams and rivers, and lakes and dams. This 'blue' water resource is used for irrigated agriculture which produces some 40% of global food needs. The rest of the food comes from rainfed agriculture based on 'green' water that is stored as moisture in the soil (Falkenmark and Rockstorm, 2006).

Precipitation is partitioned into metaphorical blue water resource which is accumulated in volumes on the surface or below ground in aquifers, wetlands, ponds, lakes, reservoirs and dams that can be exploited as source for gravity or pumped irrigation or other uses. This is distinguished from green water resource which refers to water absorbed in and stored in the soil in the unsaturated soil moisture zone or vegetation, which can be utilized by plants and other organisms through ecobiological processes. Green water flows from terrestrial biomass producing systems (crops, forests, grasslands and savannas) and blue water flows in rivers, through wetlands and base flow from groundwater (Figure 6.1).

Both blue and green water are integral and complementary parts of the water cycle in which blue water is used for irrigated agriculture where it becomes green water in the soil before being used productively for biomass production through transpiration. In rainfed agriculture, green water in the soil comes from precipitation, and is used for biomass growth through transpiration. In both rainfed and irrigated agriculture, there is a non-productive water use of green water which is water evaporation from the soil. Thus, the way the land cover is managed, and the way soil and water resources are used for agriculture has an impact on the water balance and water resource flows in the water cycle, as well as on water use efficiency, water productivity and water-related ecosystem services including the quality and quantity of water resources.

The section above has outlined how the water cycle is embedded in the natural infrastructure, and how agriculture within the natural infrastructure relates to the water cycle. Section 6.2 discusses how agricultural land use can cause the degradation of soils and water-related ecosystem services. Section 6.3 elaborates on how agricultural practices can affect the soil water balance and its components. This is followed in Section 6.4 with examples of how water-related ecosystem services can be mobilised from agricultural land. Section 6.5 deals with social and economic value of water services to agriculture and beyond, including the interactions between carbon and water cycles (Section 6.6). Section 6.7 examines the global and regional policy issues, followed the identification of options for improved management and effective development (Section 6.8). Section 6.9 deals with policy recommendations, and Section 6.10 with knowledge gaps.

6.2. Extent of agricultural systems and the degradation of soils and water-related ecosystem services

All agricultural land or agro-ecosystems have been derived originally from natural ecosystems in humid, sub-humid, or semi-arid environments in the tropics, sub-tropics and temperate climates. In ecosystems that are unsuitable for conversion to crop production, permanent rangeland systems of various types exist, including pastoral or nomadic types for animal production. In forest ecosystems, agriculture is normally based on perennial crops and on extractive uses of forests and trees. In arid and semi-arid regions where blue water resources are available for agriculture, irrigated agro-ecosystems are a common occurrence. Otherwise, agriculture relies mainly on green water for its rainfed production. Both rainfed and irrigated agriculture are altered ecosystems in which the water fluxes over a landscape or watershed can be affected by agricultural practices and these in turn affect the performance of crops as well as the water resources available for irrigated agriculture and other uses.

In natural ecosystems in which the soil has vegetation cover and the soils are biologically diverse and healthy, with high organic content and biota (including plant roots), and good porosity and structure to allow deep drainage, we find maximum infiltration and soil moisture storage in the root zone are 'normal', and the water cycle manifests minimum surface run-off. However, most soils in all agro-ecosystems today are to some extent degraded in their physical, chemical, biological, and hydrological properties compared with their undisturbed natural state. Given the multi-dimensional nature of soil systems, degradation is not a binary concept, but a matter of degree along the different dimensions, with functional consequences.

The main reason for this is the practice of mechanical tillage over many years, either with traditional implements such as the hoe or the ancient animal-drawn wooden 'ard' plough, or with modern tillage equipment such as disc, chisel and mouldboard ploughs. The main consequence of mechanical tillage combined with a lack of organic input has been to cause most agricultural soils today have low levels of organic matter with poor soil aggregate structure and low soil biodiversity. This leads in consequence to a disruption of the continuous macro-pores in the soil, for example that result from earthworm activity or root channels. These serve as drainage channels particularly beyond the tilled horizon into the subsoil. Tillage also leads to pulverization of the soil aggregates on the surface, which contributes to soil crusting and sealing thus impeding infiltration even further as illustrated in Figure 6.2 (Thierfelder *et al*., 2005).

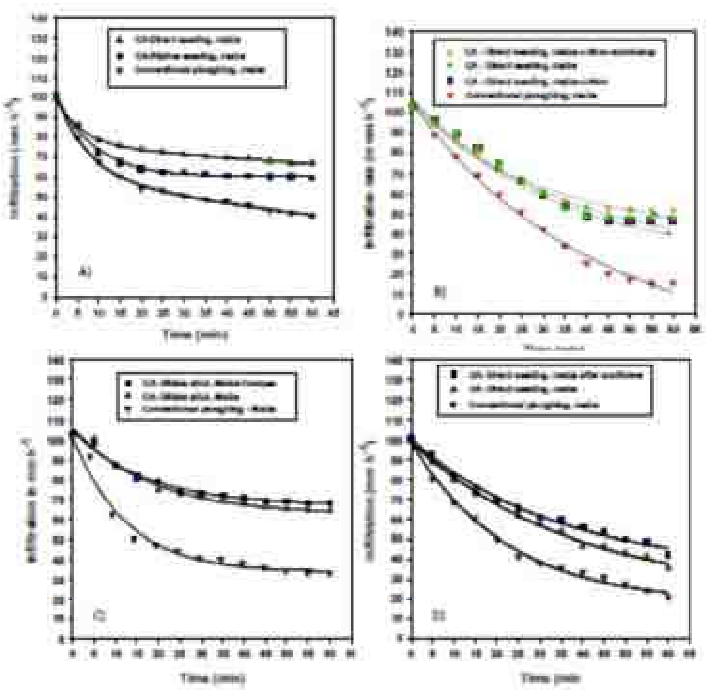


Figure 6.2. Infiltration rate in mm h^{-1} measured with a rainfall simulator in different CA and one conventionally tilled treatment at Henderson Research Station, Zimbabwe (A), Monze, Farmer Training Centre (B), Chitedze Research Station (C) and Sussundenga Research Station (D), January 2010 (Thierfelder, 2011, unpublished data).

Put simply, soil tillage and experiences with it are in fact a demonstration in practice of how the “natural infrastructure” of soils/land cover is disrupted – with subsequent loss of functions. In other words tillage, lack of soil cover and organic matter and biodiversity demonstrates the importance of natural infrastructure whose normal functions become apparent through their loss when the field is disturbed by tillage every crop season. On the other hand, tillage has a huge societal meaning for people living on agriculture. In some parts of the world, and Africa is one of them, the meaning of tillage and crop production are identical, which makes the abandonment of tillage so challenging.

Even the so called ‘normal’ agricultural soils today are less water-absorptive and water-retentive than previously, and there is widespread water erosion and attendant wind erosion. In the long run, tillage as a predominant agricultural land use practice in combination with exposed or uncovered soil surfaces leads to severe degradation and finally to desertification, as can be observed in agro-ecosystems in most parts of the world (Oldemen *et al*., 1991; Montgomery, 2007; CA, 2007; FAO, 2011).

Resulting soil erosion as well as leaching of nutrients and pollutants have lead to increased chemical and biological contamination of ground and surface water bodies, as well as sediments, leading to eutrophication and adverse effects on aquatic fauna and flora including their biodiversity. The situation regarding water security is made even worse in the arid and semi-arid areas in the tropics, sub-tropics and temperate ecosystems where tillage farming combines with negative annual water balances, short and variable rainy seasons, and increasing soil salinity to degrade soil systems more rapidly..

6.2.1 Global extents of agricultural systems

The world's agricultural area has grown by 12% over the past 50 years to 1,527 M ha of which 1,226 M ha is rainfed and 397 M ha is irrigated. In this time, population has grown from 2.5 billion to 6.8 billion, an increase of 172%, Agricultural land per capita in this time has declined from 0.545 ha per capita to 0.225 ha, a diminution of 60%. The global irrigated area has doubled over the same period, accounting for most of the net increase in cultivated land, but this remains a minority of the area cultivated. Over the next 40 years, global food output will need to increase by 70% to meet demand, but agricultural area is expected to grow by only 10% at most (FAO, 2009). This refers only to the potentially available land with some ecological suitability for agriculture, without considering natural reserves or marginal lands. However, it does not factor in the loss of productive land occurring at the same time and therefore the conclusion usually is that crop land area will not increase in the future, and over the last 3 years global cropland is actually decreasing in net terms (see: www.landcommodities.com). Thus, what is expected is a more intensive use of water calling for increased agricultural water use efficiency as well as greater water productivity (more crop per drop), along with high land productivity (more yield per hectare), nutrient productivity, and labour productivity. In other words, future intensification of agricultural output must be based on producing 'more from less' because area expansion will not be an option in the longer-run nor will be the increase application of purchased production inputs of agrochemicals and energy (FAO 2011).

For the past 100 years, or more, most increases in agricultural output have come from more extensive production -- mechanized, large-scale operations that are high-tillage and soil-health- and water-cycle-abusive. This needs to be reversed, so just thinking in terms of 'more crop per drop,' or 'more yield per hectare' misses the point that the need is not to be just squeezing more productivity out of land and water resources with present modes of production. This implies that 'business as usual' thinking can succeed, doing 'more of the same.' The implication we draw from these numbers and trends is that there needs to be a reversal in agricultural doctrine and technology, moving from extensive agriculture to sustainable intensification.

This changes the meaning of the term 'intensification,' which has previously referred to or implied intensification of *external inputs*, particularly agrochemicals (Reichardt *et al*., 1998). Here we mean intensification of *management and of output*, referring to timing, spacing, complementarities of crops and animals, etc. capitalizing on synergies within agro-ecosystems, within and among species, even within and among kingdoms of species, taxonomically speaking, to achieve the highest possible output.

Increases in agriculture productivity in the developed world and in the developing world after the green revolution has focused to a large extend on chemically induced production increases (e.g., more fertilizer – more crops). Over time this has led to negative side-effects such as pollution of surface water (nitrates and phosphates) and salinization of groundwater aquifers, particularly in the semi-arid areas. The dwindling capacity of soils to store plant essential nutrients puts agriculture production at stake. A paradigm shift is therefore necessary that has to focus more on soil health and fertility increases instead of just nutrient-induced productivity increases.

The direction of agricultural development is to make cultivation less, rather than more, dependent on chemical inputs, given their adverse effects on soil and water quality, and particularly on the soil biota, whose contribution to agricultural productivity is only beginning to be scientifically understood (Chi *et al*., 2005; Rodriguez *et al*., 2009; Uphoff *et al*., 2012)

Agriculture is the largest user and polluter of water in rainfed and irrigated agriculture, and its share of blue water for irrigated production will decrease further in the future. It is important that agricultural stakeholders all understand the root causes of soil, water and biodiversity degradation, and therefore to appreciate and prevent the degradation of the natural infrastructure, in their agricultural land use. Otherwise, it would be difficult to define and to take action on what needs to change in agriculture practices and landscape management so as to sustain/restore this 'natural infrastructure' and to improve productivity of land and its water-related ecosystem services.

6.2.2 Degradation of natural infrastructure and causes of soil and water degradation

According to the Global Assessment of Human-Induced Soil Degradation (GLASOD), all the world's agricultural land has been degraded to some degree, and half the land areas degradation has reached alarming proportion (Oldeman *et al*., 1991). Degradation of cropland was most extensive in Africa, with 65% of cropland areas affected, compared with 51% in Latin America and 38% in Asia (CA, 2007). Loss of organic matter and physical degradation of soil not only reduces nutrient availability but has also significant negative effects on: soil systems' infiltration and porosity that in turn affect local and regional water productivity, the resilience of agro-ecosystems, and global carbon cycles (Vlek *et al*., 2008).

Accelerated on-farm soil erosion leads to substantial yield losses and contributes to downstream sedimentation and the degradation of water bodies, a major reason of investments in water and irrigation infrastructure losing capacity faster than planned. Nutrient depletion and chemical degradation of soil are primary causes of decreasing crop yields, and cereal crop yields have been decreasing since the mid-80s (or mid-90s depending on how one calculates this) and they result in low on-site water productivity and in off-site water pollution. In irrigated areas, secondary salinization and water logging threaten past productivity gains (CA, 2007).

Globally, agriculture is the main contributor to both point and non-point-source water pollution. Water quality problems can often be as severe as those of water availability, but have yet to receive as much attention. Global net outflows of dissolved inorganic nitrogen to the oceans have been estimated at 18,300 tons. The European Nitrogen Assessment report from 2011 shows the economic cost in Europe from excess N applications to be 70 to 320 billion Euros. According to the Millennium Ecosystem Assessment (MEA 2005) some two-thirds of our ecosystems are degrading, some in the process of severe degradation. More recently, according to FAO (2011), only some 10% of the global agricultural land is considered to be under improving condition, with the rest suffering from some degree of degradation, and with 70% being characterized as being moderately to highly degraded.

6.3. Effect of agricultural practices on the soil water balance and on its components within natural infrastructure

It is now evident from international scientific studies and from observational evidence from farmers' fields and agricultural landscapes that the tillage agriculture which pulverises and exposes soils and destroys soil life and health (that has been promoted in the US after WWI, as soon as 'tractorization' became possible, and since the end of WWII more generally in the North and continues to be promoted in the South) is unable to deliver the water-related ecosystem services adequately and has high negative externalities (Pretty, 2008; Kassam *et al*., 2009; Wezel *et al*., 2009; FAO, 2011).

Thus, a fundamental question regarding water security for agriculture is: how to manage the soil-water-biological resources of the natural infrastructure to produce more with the same or even lesser amounts of water (and other production inputs) available from rainfall and irrigation, from surface or sub-surface sources, and do so with minimum negative externalities? For water, this question is linked to the possibilities of minimizing at least unproductive on-field water losses, namely run-off, soil evaporation and deep percolation beyond the root zone. From a water cycle perspective, all these components are also important to replenish bluewater resources. From an agronomist's viewpoint, reductions of these losses must occur not only to achieve the goal of having higher water productivity (WP), and water use efficiency (WUE) or, as suggested by Blum (2009), more efficient water use

(EWU), but also because run-off losses, if uncontrolled, can have other severe and harmful consequences such as pollution of water with agro-chemicals, pesticides, microorganisms and sediments.

Considering the limitations on suitable land and water resources to produce enough commodities for the future world population, while sustaining other ecosystem services provided by agriculture, one of the main strategies to deal with the large trade-offs between water uses is improving water management practices on agricultural lands (Gordon *et al*. 2010). This requires a careful look at the water fluxes described in Figure 6.1 and examination of the processes and factors that affect soil water, as this is ultimately the source of green water for plant growth and biomass production, and of all the biological products from agriculture, and also the ground water and stream flow which is the source of blue water for irrigation production, and for meeting other water needs.

In agriculture, soil management and production practices must be such that all the water-related processes of the soil water balance, such as infiltration, run-off, soil evaporation, transpiration, soil water retention, and deep drainage, maintain a favourable soil moisture status for crop production, without detrimentally altering the water fluxes in the hydrological cycle. Figure 6.1 shows that soil water status is linked to the above-mentioned water processes and there is a set of factors that can affect each of the water processes. Of all the farming operations, soil tillage, soil mulch cover and residue management, but also crop and, eventually irrigation management have a major direct impact on the water cycle and on water security. These are addressed in this section.

6.3.1 Hydrological functions of "Natural Infrastructure" of soils and land cover as demonstrated through the impacts of Soil Tillage

The objectives of soil tillage are manifold, and the most popular arguments for it include weed control, soil loosening and decompaction, seed bed preparation, crop residue management. Based on the processes such as infiltration, runoff, and transpiration that affect soil water status, outlined in Figure 6.1, this section reviews the effects of soil tillage practices, soil cover, and residue management on these processes and the consequences on crop-available soil water and its use efficiency.

6.3.1.1 Effects on infiltration and run-off

Tillage practices normally affect infiltration behaviour of soils, and thus run-off, as a consequence of the modification of soil properties such as stability of structural soil aggregates, total and macro-porosity, hydraulic conductivity, surface crusting and compaction, and soil organic matter. Generally, soil aggregation improves when farmers convert from conventional soil tillage to no-tillage soil management. As a result, greater pore connectivity takes place, enhancing soil quality and water transmission properties.

Despite the occasional need for tillage operations to enhance infiltration, limited e.g. by surface crusts or compacted soil layers (often as result of little soil cover (Thierfelder and Wall, 2009; Singh *et al*., 2005) or repeated ploughing to a given depth, (Reichert *et al*., 2009~~b~~; Mary and Changying, 2008; Sasal *et al*., 2006; Hamza and Anderson, 2005)), tillage seems to have only a short-lasting effect on the improvement of the infiltration rate (Freese *et al*., 1993). In fact, soil disturbance caused by tillage operations does increase surface roughness, macroporosity, and initial infiltrability of the soil, but the infiltration rate rapidly declines over time as a consequence of aggregate collapse (Guzha 2004). Although soil roughness effectively retards run-off and increases the time available for infiltration, in warm and dry environments the water that ponds on the soil surface quickly evaporates, thus reducing total infiltration and effective rainfall (Peterson and Westfall, 2004).

In terms of water infiltration, several research studies have highlighted significant advantages of no-tillage systems over conventional tillage practices (Photo 6.1). Numerous studies from throughout the world confirm that no-tillage systems promote soil aggregation accompanied by improved water infiltration. In Brazil, Calegari *et al*., (1998) found a 2.5 times greater infiltration rate under NT when compared to conventional tillage, and Stone and Schlegel (2010) observed a 2.67 times improved infiltration rate comparing the same soil management systems. From several rainfall simulator studies Thierfelder (2011, unpublished data) reports consistent greater steady-state infiltration rates under no-tillage attributed frequently to the presence and stability of surface-connected macropores, but also a higher concentration of larger, water-stable aggregates in the surface layer, and reduced surface sealing.

Photo 6.1 Soil compaction and loss in water infiltration ability caused by regular soil tillage leads to impeded drainage and flooding after a thunder storm in the ploughed field (right) and no flooding in the no-till field (left). Photograph taken in June 2004 in a plot from a long-term field trial "Oberacker" at Zollikofen close to Berne, Switzerland, started in 1994 by SWISS NO-TILL. The three water filled "cavities" in the no-till field derive from soil samples taken for "spade tests" prior to the thunder storm (credit: Wolfgang Sturny).

Macroporosity, understood as the fast draining air-filled porosity (>50 µm), and its geometry plays a key role for infiltration and redistribution of soil water in depth (Sasal *et al*., 2006). Especially surface connected, vertically oriented biopores as created by mesofauna, mainly earthworms (Kladivko *et al*., 1986), or by roots after their decomposition contribute to high infiltration rates (Imhoff *et al*., 2010; Buczko *et al*., 2003; Tebrügge and Abelsova, 1999), even under low total soil porosity or compacted conditions.

6.3.1.2 Effects on water retention capacity and deep drainage

Plant-available water in a given volume of soil depends on the amount of mesopores, able to retain water against gravitational forces and to deliver it to roots upon demand (50 µm > pore diameter > 0.2 µm). In addition, the soil depth to which roots are able to extract water is equally important as plants can compensate for water stress in the upper, more densely rooted soil layers by water uptake from deeper layers (Teuling *et al*., 2006). Any kind of soil disturbance, by affecting directly or indirectly the pore size distribution, pore geometry, and hydraulic properties of soil, influences plant-available soil water.

Today it is widely recognized that the absence of soil disturbance improves aggregate stability and promotes SOM accumulation and stabilization (Fernandez-Ugalde *et al*., 2009; da Veiga *et al*., 2008; Bescansa *et al*., 2006). There further exists a good correlation between soil structural stability, SOM content and plant-available water (Imhoff *et al*., 2010, Abid and Lal, 2009; So *et al*., 2009; Mrabet *et* *al*., 2001). Despite a frequently observed reduction of total porosity in the surface soil layer under no-till, the total volume of mesopores in the soil is increased in the absence of soil disturbance. After six years of differentiated tillage (no-till and moldboard plough) on a Vertisol, Carvalho and Basch (1995) found lower total and medium-size porosity under no-till in the 0-0.1 m soil layer; however in the layers between 0.1 and 0.3 m, total porosity and especially the pore space retaining plant-available water were considerably increased (Table 6.1). However, in the review of Kay and VandenBygaart (2002) on conservation tillage and depth stratification of porosity and soil organic matter, the authors identified some cases where tillage-induced changes in mesoporosity did not occur. According their interpretation, only long-term studies are able to provide consistent information, especially with regard to changes in SOM and changes in pore size fractions.

Table 6.1. Total porosity, pore size distribution, plant available water and soil organic matter content in a vertic Cambisol after 6 years under no-till (NT) and conventional tillage (CT) (Carvalho and Basch 1995)

TILLAGE	DEPTH (cm)	> 50 µm (%)	50 -10 µm (%)	10-0. 2µm (%)	< 0.2 µm (%)	total porosity (%)	available water (%)	SOM (g kg^{-1})
NT	10	3.20	2.22	2.7	38.37	46.52	4.92	2.53
	20	0.86	3.91	5.22	36.16	46.15	9.13	2.15
	30	1.86	2.63	11.48	29.44	45.4	14.11	2.25
	0-30	1.97	2.92	6.47	34.66	46.02	9.39	2.31
CT	10	15.08	2.34	4.36	29.95	51.73	6.71	1.58
	20	2.67	1.32	2.31	39.95	42.25	3.63	1.7
	30	1.47	1.56	3.29	35.62	41.94	4.85	1.66
	0-30	6.41	1.74	3.32	35.17	45.31	5.06	1.65

Chemical and physical subsoil constraints may limit water uptake from deeper soil layers (Dang *et al*., 2010; MacEwan *et al*., 2010; Nuttall and Armstrong, 2010). Water stored in deeper soil layers can play a decisive role for crop yields as it is used later in the season during grain-filling (Passioura and Angus, 2010; Kirkegaard *et al*., 2007). Deep soil loosening has become a widely-used soil management practice to overcome physical subsoil constraints, where powerful tractors and sub-soiling equipment are available (Hamza and Anderson, 2005). Wetter regions and sandy soils are more likely to benefit from subsoiling than fine textured soils and drier locations (Wong and Asseng, 2007). Even negative results from deep-loosening were obtained in dry years, or when deep drainage losses occurred (Wong and Asseng, 2007). Further, the effects of deep-loosening are often of short duration, especially if not accompanied by additional measures, such as the application of subsoil conditioners (i.e., gypsum), installation of primer crops (stabilizing deep-rooting legumes), or reduced or controlled traffic (Lopez-Fando *et al*., 2007; Hamza and Anderson, 2005; Hamza and Anderson, 2003; Yunusa and Newton, 2003).

Deep drainage occurs either when the soil volume above is saturated and the wetting front reaches the lower limit of the rooting zone, or through preferential pathways for drainage even if the soil profile is not saturated. It must be considered an important process to recharge groundwater and to conduct excess water through the soil profile to deeper soil layers, thus contributing to reduce surface run-off. However, in dry regions, deep drainage is usually regarded to correspond to a loss in crop available water (Passioura, 2006).

Tillage may affect deep drainage either through its impact on rooting depth or on the build-up and longevity of macropores. The absence of soil disturbance preserves biological macropores, earthworm tubes, former root channels, and voids between soil structural units, thus forming preferential pathways for rapid and deep percolation (Verhulst *et al*., 2010; Cullum, 2009; Strudley *et al*., 2008; Shipitalo *et al*., 2000; Tebrügge and Düring, 1999). After 18 years of no-till and conventional tillage on a Vertisol, McGarry *et al*. (2000) found that deep drainage was also enhanced under NT, besides taking a longer time to ponding, final infiltration rate, and total infiltration. The withdrawal of excess water from saturated topsoil through deep drainage, however, provides a basis for a positive rebalancing between the consequent increase in infiltration and the reduction in run-off.

6.3.1.3 Effects on soil evaporation

In arid and semi-arid regions, the main unproductive water loss is caused by direct evaporation, especially if many, low-intensity rainfall events are predominant (Lampurlanés and Cantero-Martínez, 2006; Passioura, 2006). During the initial evaporation stage, evaporation from a bare soil surface can be reduced through a coarse or disturbed layer (or mulch) overlying the wet subsoil. Frequently however, tillage operations to reduce evaporation losses are carried out at the end of the initial evaporation stage with the soil under friable conditions. Soil loosening and exposure further boosts water losses through faster heating and the formation of air pockets in which evaporation occurs (Licht and Al-Kaisi, 2005). Within 24 hours after primary soil tillage, Moret *et al*. (2006) measured up to 16 mm of evaporation losses compared with 2 mm under no-till. After secondary tillage, the authors still measured differences of up to 3 mm of water loss between tilled and untilled treatments. The objective to reduce evaporation through superficial tillage operations is therefore the result of a balance between short-term evaporation losses through the enhanced drying of the soil layer disturbed by tillage and possible long-term gains through the interruption of easy upward capillary movement in intact continuous pores as it is the case in undisturbed soil.

Evaporation from soil may also be affected by tillage through an increase in surface roughness by exposing a greater surface to the overlying atmosphere and winds, and through a change in soil surface temperature and albedo. Albeit a higher surface albedo on a smooth bare soil when compared to moldboard-ploughed soil or conventional tillage, Oguntunde *et al*. (2006) found only small differences in soil moisture content of the surface layer. However, on swelling and shrinking soils, evaporation losses may be considerable. Measuring evaporation from the cracks near the end of the sorghum-growing season, Ritchie and Adams (1974) obtained losses of 0.6 mm/day with an additional 15 mm of soil water was lost by evaporation before the following rains closed the cracks. Mulching or superficial soil tillage could prevent the formation of cracks or at least obliterate them after they have begun to form.

6.3.2 Hydrological functions of "Natural Infrastructure" of soils and land cover as demonstrated through the impacts of soil mulch and residue management

In natural ecosystems where the amount of rainfall allows the growth of some kind of vegetation, land surface develops some form of organic cover comprised of plants and their residues after senescence. This soil cover controls the flux of energy and water by interacting with components of atmosphere, hydrosphere, biosphere and pedosphere (Lal, 2009). Normally, transforming natural environments into agricultural areas leads to major changes in land cover and a significant change in the partitioning of water, nutrient, carbon and energy flow. Soil cover and residues directly affect run-off and soil evaporation and indirectly deep percolation, all of them representing unproductive water losses. The objectives of sustainable soil and soil water management are to redirect these losses into an increase of soil water storage and availability to plants. However, in certain cases (plain areas or impermeable soil layers) improved infiltration may increase the risk of water logging (Thierfelder and Wall, 2009, 2012).

This section addresses the influence of soil cover, including the application of organic and inorganic mulching material, cover crops and crop residues on soil water, either through their direct impact on infiltration/run-off and evaporation, or their indirect effects on soil organic matter content and mesofauna activity. Additionally, evidence is presented on how type of soil cover (including cover crops) and residue characteristics and their management affect soil water conservation and soil productivity.

6.3.2.1 Effects on infiltration and run-off

Soil macroaggregate breakdown is seen as the major factor leading to surface pore clogging by primary particles and microaggregates and thus to formation of surface seals or crusts (Lal and Shukla, 2004). Soil cover prevents this breakdown by reducing the kinetic energy with which raindrops reach the soil surface (Ben-Hur and Lado, 2008). In addition to the detachment of soil aggregates through direct raindrop impact and the physicochemical dispersion of clays, slaking is considered another important process in the disintegration of aggregates and the consequent seal formation (Lado *et al*., 2004).

The amount of crop residues and their management can have a decisive effect on the resilience of aggregate breakdown and on the processes of particle detachment and slaking. Several authors found a direct positive relationship between the amount of straw residues and aggregate stability (Jordan *et al*., 2010, Mulumba and Lal, 2008). Yet, it seems that crop residues alone are not effective in improving soil aggregate stability. Comparing different residue management systems, including no-till and tillage with (replacement after) and without residue removal, Wuest (2007) found a significant improvement of aggregate stability only under no-till. Naudin *et al*. (2003) also report on the beneficial effect of the permanent soil cover with a mulch to prevent the hardening of Alfisols and to increase their water reserve.

If crop residues are left on the soil surface, or are partially incorporated in the upper soil layer through mesofauna, not only is the impact of raindrops reduced but also the stream velocity, as the residues act as a succession of physical barriers (Verhulst *et al*., 2010, Jin *et al*., 2009). Residues play a role similar to that of surface roughness, i.e., increasing the time for infiltration to take place (Blevins and Frye, 1993), with the difference that their effect lasts longer. Therefore, the time lag for run-off generation is also greater when crop residue is left on the soil surface (Jordan *et al*., 2010) and the transmission losses (turning run-off at small scale into run-off at large scale) decrease with increasing vegetation or residue cover (Leys *et al*., 2010).

Thus, soil cover and crop residues left at the soil surface seem to be effective in improving infiltration and in reducing surface run-off and soil loss. It also appears that the amount of residues is closely related to the degree to which run-off is decreased. After three years under different mulching rates of wheat straw, rainfall simulation measurements at intensities of 65 mm h^{-1} provided clear differences in surface run-off between mulching rates (Figure 6.3) (Jordan *et al*., 2010). In this study, the highest straw residue rates of 10 and 15 Mg ha^{-1} were necessary to almost completely avoid run-off.

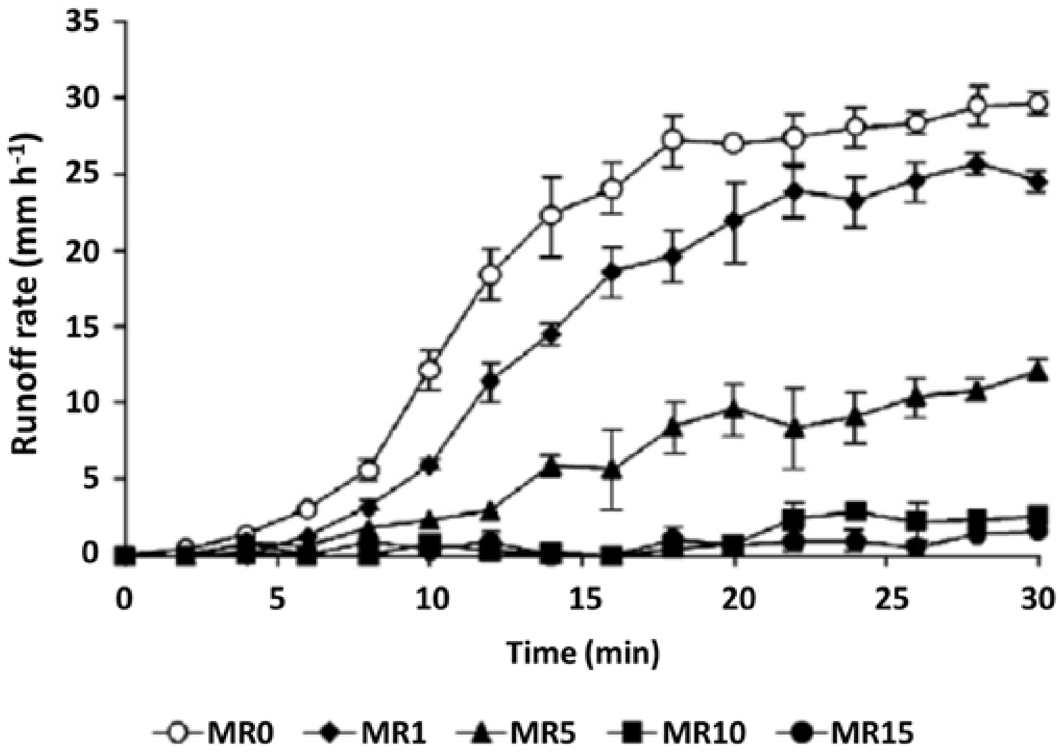


Figure 632 Variation of mean run-off rates under different mulching rates. MR0=control; MR1=1 Mg ha^{-1} year^{-1}, MR5=5 Mg ha^{-1} year^{-1}; MR10=10 Mg ha^{-1} year^{-1}; MR15=15 Mg ha^{-1} year^{-1}. N=5 for each mulching rate treatment. Vertical bars indicate ± standard deviation (Jordan et al. 2010)

A big difference between high and low standing, surface-cut and removed stubble has been found in regions where the retention of snow is crucial for supplying water to the following crop. Sharratt (2002) reported that taller stubble trapped more snow, reduced the depth of frost penetration, and hastened thawing of the soil profile by at least 25 days, when compared to short-stubble or residue removal. Further, the variability in soil water recharge was closely related to the amount of snow cover.

6.3.2.2 Effects on evaporation

The processes that govern evaporation from soil and which are affected by soil mulch cover and residue management are the reflection of incident energy reducing energy absorption by the surface, the lowering of wind speed at the soil surface, and the strong reduction of the vapor flux from soil into the atmosphere.

Soils mulched with crop residues or cover crops have reduced maximum soil temperature and lower amplitude (Zhang *et al*., 2009, da Silva *et al*., 2006, Fabrizzi *et al*., 2005), due to their higher solar reflectivity and lower thermal conductivity (Shinners *et al*., 1994, Hillel 1980). Trevisan *et al*. (2002) showed a reduction in soil temperature amplitude down to 20 cm depth with oat straw cover throughout the year compared with soil without straw cover. In addition, soil evaporation is negatively correlated to the amount of crop residues left on the soil surface, irrespectively of the evaporative demand of the environment (Table 6.2, Freitas *et al*., 2006).

Table 6.2. Total soil evaporation during 21 days (after reaching field capacity) for two soils under different types and amounts of residues and different evaporative demands (adapted from Freitas et al. 2006)

			Evaporative demand (mm d^{-1})					
	Residues (kg ha^{-1})		Corn			Wheat		
Soil type	Corn	Wheat	8	6	3	7	5.2	3
Loamy sand	0	0	74.2	82.0	57.2	59.2	68.0	47.9
Heavy clay	0	0	56.4	74.2	56.4	54.7	59.0	46.9
Loamy sand	5000	3500	40.2	28.9	19.0	38.0	28.4	18.5
Heavy clay	5000	3500	35.7	30.1	22.2	35.2	32.0	22.8
Loamy sand	10000	7000	20.4	19.8	18.6	20.6	20.0	16.5
Heavy clay	10000	7000	21.1	18.1	13.6	20.3	17.1	13.1

However, residue management effects on evaporative water losses may vary with site-specific conditions. For example, Steiner (1989) found that residue thickness (volume) is more important than mass per unit area for controlling evaporation. Under sandy topsoil conditions Ward *et al*. (2009) observed an increased evaporation in the presence of standing stubble when compared to cut and removed or slightly buried stems, whereas Klocke *et al*. (2009) found a higher evaporation reduction effect of standing wheat straw compared to flat corn residue. For Sillon *et al*. (2003) and Gill and Jalota (1996) the efficient break of the unsaturated hydraulic conductivity through surface incorporated residues is a more effective way to reduce evaporation losses than the retention of a sufficient amount of residues. However, under some climatic conditions any incorporation of residues into the soil can boost the decomposition to an extent that the soil organic matter levels are overall reduced (Sa *et al*. 2008), apart from the disruption of the macropore structure by the tillage operation.

6.2.3 Effects on soil water through the increase of soil organic matter and mesofauna activity

The removal of crop residues through burning or for fodder and biofuel purposes is considered to be a major threat to soil productivity, environmental quality, and overall sustainable development (Blanco-Canqui and Lal, 2009; Hakala *et al*., 2009; Lal, 2009). Besides giving physical protection to the surface soil layer and having an impact on infiltration and evaporation, organic residues enhance the build-up of soil organic matter and soil fauna activity which contribute to improving soil porosity, soil particle aggregation, soil moisture storage, and deep water infiltration (Lal, 2009; Wuest *et al*., 2005).

The improved pore space is a consequence of the bioturbation activities of earthworms and other macro-organisms and channels left in the soil by decayed plant roots. Studying the effects of earthworms in Germany, Ernst *et al*. (2009) found that the soil water was strongly affected by the activity of ecologically different earthworm species. The epigeic *Lumbricus rubellus* tended to enhance the storage of soil moisture in the topsoil, and the endogeic *Aporrectodea caliginosa* strongly improved water infiltration and hastened the water discharge through the soil. Although the benefits of increased earthworm populations are mainly attributed to the absence of soil disturbance (2–9 times more in no-till than under conventional tillage (Chan, 2001) and relatively less to the amount of residues retained at the soil surface (Eriksen-Hamel *et al*., 2009), Blanco-Canqui and Lal (2007~~b~~) found a strong effect of corn stover removal on the reduction in the number of earthworms.

Soil organic matter promotes soil biological activities and processes, resulting in more bacterial waste products, organic gels, fungal hyphae (polysaccharides), and worm secretions and casts (Wuest *et al*., 2005), which improve aggregate stability and porosity. Directly or indirectly, these organic compounds are related with water-holding capacity, although it is the total soil organic carbon or organic matter that is usually considered as an important aggregate indicator in the discussion on water-retention pedofunctions (Rawls *et al*., 2003). Evaluating the efficiency of pedotransfer functions to estimate water retention in 725 soil samples from Rio Grande do Sul State, Brazil, covering all types of soil textures, Reichert *et al*. (2009~~a~~) concluded that organic matter must be included as an independent variable, because it had an individual positive effect on field capacity and plant-available water.

Crop residue incorporation is not the best residue management practice because it implies soil disruption and eliminates the beneficial effects of residues retained on the soil surface. Even so, in a long-term experiment, Singh *et al*. (2005) found that rice-straw incorporation was less detrimental to soil physical and hydraulic properties than the burned or removed rice straw. Whereas straw removal compared to the other residue management systems performed worst with regard to soil organic matter and soil aggregation, straw burning led to reduced water retention due to an increased water repellency of the soil surfaces.

6.2.4 Influence of the type of soil cover and residues and their management on soil water and crop productivity

As shown in the previous sections, soil cover has a decisive effect on soil water dynamics and contributes to enhance the green water component and promote water productivity. The possibilities and the choice of the soil cover and its impact depend, besides the main objective behind it (e.g. soil structure improvement, nitrogen fixation) and the possible uses for human and/or animal consumption, on climate, hence length of the growing season, soil properties, management and cropping systems (the cover crop grown in association is chosen according to the main crop, usually cereal - legume associations), farmer’s investment capacity and access to market (Wilhelm *et al*., 2004) and alternative uses of biomass (most important the competition for the limited biomass between mulching and forage production in the case of integrated crop-livestock production systems and remunerative crop residues) (Lal, 2009), among others.

In general terms, soil cover can be achieved locally either using residues from the previous crop or a cover crop in the crop rotation. As an alternative, a cover crop could be kept alive and controlled during the main crop cycle (intercrop); this would be the most efficient alternative but it requires high technical skills. Whereas the use of cover crops is mainly restricted to humid or sub-humid regions, in semi-arid environments the cover crop can compete, in time or space, with the main crop and soil cover through crop residues is the most commonly used option to improve the use efficiency of the main limiting factor to crop productivity.

Although in dry areas the short rainy (hence growing) season may not allow the production of a cover crop on a yearly basis before, after or during the main crop, it is necessary to produce an important biomass the preceding year. In such ecologies, it is capital to use all the available water and benefit from the high mineralization at the beginning of the rainy season. Early sowing, immediately after the first useful rain, is made possible by direct seeding techniques when mulch is prepared in the dry season.

On a very limited scale other materials such as plastic films, gravel or sand, or organic waste products may be an option for mulching to protect the soil and enhance the green water component. The use and the effectiveness of these materials have been reported mainly from Asian countries, and are considered an option for reducing soil evaporation, increasing infiltration of rainwater and soil water retention (Liu *et al*., 2010; Ghosh *et al*., 2006; Ramakrishna *et al*., 2006).

In large-scale agriculture and especially under semi-arid conditions, crop residues including from cover crops seem to be the only technically-feasible and economically-viable option to cover and protect the soil, while improving soil water and water use efficiency. However, studies that relate long-term residue cover to soil water availability and crop productivity under field conditions are scarce and sometimes inconclusive or contradictory (Blanco-Canqui *et al*., 2006) as the benefits of residue cover in terms of soil fertility and water availability might be offset by locally or regionally relevant problems such as poorly-drained soils and sub-optimal springtime temperatures (Lal, 2008~~a~~; Fabrizzi *et al*., 2005; Anken *et al*., 2004), weed and pest problems (Ngwira *et al*., 2012; Mann *et al*., 2002) or soil nitrogen lockup (Gao and Li, 2005).

The problem of competition for residues in crop-livestock farming systems is well known (Kassam *et al*., 2012, Valbuena *et al*., 2012), and the increasing demand of residues for biofuel production (Graham *et al*., 2007; Wilhelm *et al*., 2004) is also raising concerns regarding excessive residue removal (Lal, 2009). Dabney *et al*. (2004) alert that the benefits of long-term no-till management may be lost by the excessive removal of crop residues. Studying different percentages of corn stover removal over two years on three long-term no-tilled sites in Ohio, Blanco-Canqui and Lal (2007~~b~~) found a decrease of plant-available water with the increase of percent stover removal. However, this was reflected in higher crop yields only at one site, well drained but erosion-prone. They concluded that soils with different characteristics might reveal yields effects if stover removal above a certain threshold level was continued over a longer time period, and further that site-specific determinations of these threshold levels were urgently needed. However, these thresholds should also be assessed with regards to other ecosystem services provided by retaining crop residues, such as off-setting CO_2 emissions and maintaining the overall soil quality (Lal, 2005) or facilitating weed management and hence reducing the use of herbicides.

Cover crops are grown for multiple reasons and their use may present advantages and disadvantages as reviewed by Dabney *et al*. (2001). With regard to soil moisture conditions for the main crop, the benefits may derive from higher water infiltration, less evaporation losses through increased residue cover, increase in soil organic carbon, improved soil physical properties (Lu *et al*., 2000), or removal of excess water from a wet soil to allow timely establishment of the next crop (Unger and Vigil, 1998). However, the reduction of soil moisture is the main reason why cover crops are more suited to sub-humid and humid regions, unless irrigation is available to compensate the extra water consumption by the cover crop. The use and the choice of cover crop species is highly site-specific and depends on the main objective to be achieved. Short-cycle and early-maturing species or premature interruption of the cover crop cycle have been proposed to reduce competition with the main crop (Whish *et al*., 2009; Salako and Tian, 2003; Zhu *et al*., 1991). In semi-arid regions with summer or winter rainfall, normally a single cash or food crop is produced during the growing season often followed by fallow. In some regions, more than a third of the agricultural land may be under fallow. With no-till system of soil and crop management, it has been shown that introduction of cover crops (for forage or grain) in rotation can reduce fallow land and simultaneously improve soil cover, rainwater infiltration, soil water storage, biological nitrogen fixation (in case of legumes), and soil organic matter and fertility (Goddard *et al*., 2008, Crabtree, 2010), while reducing soil evaporation as already indicated from crop residues (Jalota and Arora, 2002). This has been shown to work in semi-arid regions in many parts of the world including North Africa (Mrabet, 2008); Canada (Baig and Gamache, 2009; Lindwall and Sonntag, 2010); USA (Ransom *et al*., 2007), Australia (Flower *et al*., 2008), and Eurasia (Gan *et al*., 2008). Similarly, with irrigated systems, off-season cover crops provide similar advantages.

6.4 Evidence for hydrological functions and services: mobilizing water-related ecosystem services from agricultural land

Three cases are presented in this section to illustrate the positive impact that agriculture production systems can have on the water-cycle components, and thus contributing to water security. These are: (1) Conservation Agriculture (CA), an approach to sustainable production intensification, which is now spread across all continents and in all ecologies, covering some 125 million hectares; (2) the Itaipu watershed services in the Paraná basin III in Brazil based on CA that has allowed the reduction of soil erosion and the delivery of clean water to the Itaipu dam for generating hydro-electric power for Brazil, Argentina and Paraguay; and (3) the System of Rice Intensification (SRI) which is a an alternate way of producing irrigated or rainfed flooded rice with up to 50% lower water requirement.

6.4.1 Conservation Agriculture (CA) – An Approach to Sustainable Production Intensification and Water-related Ecosystem Services

Soil conservation measures were developed after the North American dust bowl disaster in the 1930s. The first measures involved practices such as contour ploughing, terracing and/or strip cropping to reduce run-off and soil erosion. According to Derpsch (2004), research on reduced tillage with early versions of a chisel plough was initiated in the Great Plains in the 1930s to alleviate wind erosion. Stubble-mulch farming was also developed and can be seen as a forerunner of no-tillage farming. This collection of practices led to what became more widely known as conservation tillage, which includes a range of tillage practices from low soil disturbance to high soil disturbance, with soil at least 30% of the soil covered with crop residues.

The book *Ploughman's Folly* by Edward Faulkner (1945), in which he questioned the wisdom of ploughing and explained the damaging nature of soil tillage, was an important milestone in the development of conservation agricultural practices. Further research in the UK, USA and elsewhere during the late-1940s and 1950s, and the development of herbicide technology, made no-till farming more viable economically, and the practice began to spread in the USA in the 1960s, and in Brazil, Argentina, Paraguay and Australia in the 1970s. In 1973, Shirley Phillips and Harry Young published the book *No-tillage Farming*, the first of its kind in the world, and this was followed in 1984 by the book *No-Tillage Agriculture: Principles and Practices* by E.R. Phillips and S.H. Phillips: see references.

The modern successor of no-till farming – generally known now as Conservation Agriculture (CA) – goes much further. It involves the simultaneous application of three practical principles based on locally-formulated practices (Hobbs, 2007; Friedrich *et al*., 2009; Kassam *et al*., 2011a): minimising soil disturbance (no-till seeding); maintaining a continuous soil cover of organic mulch and plants (main crops, residues and cover crops including legumes); and cultivation of diverse plant species that, in different farming systems, can include annual or perennial crops, trees, shrubs and pastures in associations, sequences or rotations, all contributing to enhance system resilience, in conjunction with good crop, nutrient, weed and water management. This is the core of FAO's new agricultural intensification strategy (FAO, 2011).

World-wide, CA is now practised on an estimated 125 million ha, mainly in North and South America, and in Australia, but also increasingly in China, Kazakhstan, Ukraine and Russia (Table 6.3). During the past decade, it has begun to spread in Asia more generally (including on the Indo-Gangetic Plains), in Europe (including in the UK) as well as in Africa. CA has now spread over 1 million ha in Africa, including in South Africa, Mozambique, Malawi, Zambia, Zimbabwe, Madagascar, Kenya, Sudan, Ghana, Tunisia and Morocco, and some two-thirds of the area is under small-holder production. Much of the latter adoption has occurred in the past 4 to 5 years as a result of more extension attention and development resources being directed towards the promotion of CA through participatory dissemination approaches.

Table 6.3: Global Adoption of Conservation Agriculture (Source: Friedrich, Derpsch and Kassam (2012))

Continent	Area (ha)	Percent of total
South America	55,464,100	45
North America	39,981,000	32
Australia & New Zealand	17,162,000	14
Asia	4,723,000	4
Russia & Ukraine	5,100,000	3
Europe	1,351,900	1
Africa	1,012,840	1
World total	**124,794,80**	100

No-till farming was introduced as a means to control soil erosion and sustain crop production on erodible or degraded soils, and it has been promoted mainly for that purpose. However, during the past decade, CA has become the flagship of an alternative agricultural paradigm for intensifying crop production that not only improves and sustains productivity but also delivers important environmental services (Kassam *et al*., 2009, 2011a; FAO, 2011). The elimination or minimisation of mechanical soil disturbance avoids or reduces the shattering of topsoil structure and pores, loss of soil organic matter, and soil compaction which occur with tillage which contributes to decreased infiltration of water and increased waterlogging (Photo 1), run-off and soil erosion, decreased soil moisture-holding capacity and less rooting volume, and degradation of soil health and productive capacity.

Maintaining a continuous cover of plants and organic mulch protects the soil against the direct impact of rain drops, enables more rainwater to enter the soil, and eliminates evaporation of moisture from bare soil. The build-up of soil organic matter from plant residues left on the soil surface – aided also by their protecting the soil surface against desiccating hot sunshine and wind – improves soil structure and porosity which, in turn, increase soil moisture absorption and storage capacities. Covering soil with organic mulch also increases the numbers of soil micro-organisms and meso-fauna, particularly earthworms, that help to break down plant remains and incorporate them in the soil, make nutrients available to plant roots, and create biopores of various sizes that improve both soil water-holding capacity and soil drainage. The use of deep-rooting leguminous crops in rotations or as intercrops can further increase soil porosity as well as provide free nitrogen to soils.

In addition to the above economic benefits, which may not always be easy for small farmer to harness (Wall, 2007), CA also provides considerable environmental benefits (Kassam *et al*., 2011). Not only does CA prevent soil erosion and help to bring degraded soils back into production, but it can also greatly reduce deforestation and burning of savannah vegetation in areas where shifting cultivation is practised. Integrating CA practices into shifting agriculture, for example, can help to transform it from 'slash and burn' farming to a 'slash and mulch' system, with potential for enhancing soil productive capacity and agricultural production over time. However, where land is not in short supply, due to benefits only becoming evident after several seasons, there is often no interest in adopting CA.
In areas of intensive agriculture, CA greatly reduces or eliminates chemical pollution of rivers and groundwater caused by fertilizer run-off and leaching that can occur under customary intensive practices. It also reduces emissions of carbon dioxide, methane and nitrous oxide to the atmosphere as reported by Parkin and Kaspar (2006), Baig and Gamache (2009) and Ceja-Navarro *et al*. (2010). Further, by increasing soil organic matter contents, it increases carbon sequestration (West and Post, 2002; CTIC/FAO, 2008; Reicosky, 2008; Baig and Gamache, 2009).

6.4.2 Watershed Services

The partnership programme established in Paraná III Basin in Brazil is considered here as a good example of ecosystem management for the provision of watershed services at the Basin level.

As part of a strategy for improvement, conservation and sustainable use of natural resources, the Itaipú Dam *Programa Cultivando Água Boa*(cultivating good water), has established a partnership with farmers to achieve their goals in the Paraná III Basin located in the western part of Paraná State on the Paraguay's border (ITAIPU 2011; Mello and van Raij, 2006). The dam´s reservoir depends for long-term productivity on the sustainable use of soil and water in the watershed for efficient electricity generation. Sediments and nutrients entering the reservoir resulting from inappropriate land use pollute the water used by the turbines to generate electricity. This phenomenon shortens the reservoir life´s and increasesthe maintenance costs of power-generating turbines, thereby increasing electricity generation costs. Thus, in principle, payments can be justified and could be made through a programme to improve the conditions of electricity generation.

The spatial unit in this programme is the watershed. If the farmers in the watershed function as a community, they can reach a scale where environmental impact can be monitored with suitable indicators to establish a system for payment for environmental services. Overall, as result of the control of erosion and a reduced sedimentation load in the water flowing into the reservoir, the life of the dam complex has been increased from its original estimate of some 60 years when the dam was built to some 350 year now.

One of the partnerships built in the *Cultivando Água Boa* programme was developed through an agreement with the Brazilian No-till Federation (FEBRAPDP) is the Participatory Methodology for Conservation Agriculture Assessment Quality (Laurent *et al.*, 2011). Through this programme, at first, the partners plan to start by measuring the impacts of farm management through a scoring system which indicates how much each farm is contributing to improving the water conditions. The system is available online in Portuguese at: http://plantio.hidroinformatica.org/.

Consolidating this phase and adapting the principles established for the 'water producer' by the National Water Agency, the partners plan to assign values to ecosystem services generated from farms participating in the programme (ANA, 2011). Considering the polluter/payer and provider/receiver principles set in the Brazilian Water Resources Policy, farmers with good scores will be paid for their proactive action to deliver watershed servicesonce the Paraná Watershed Plan is established. This will be a new framework for services provided by farmers as compensation for their proactive approach to improve the reservoir water quality and reducing costs for electricity generation by the Itaipú Dam.

6.4.3 The System of Rice Intensification (SRI): More Productivity with Less Water

The System of Rice Intensification (SRI) – a rice production system based on modifying standard crop and water management practices, rather than on introducing new varieties or on using more purchased agrochemical inputs -- has taken root on an international scale, moving far beyond its origins in Madagascar. For our discussion here, most significantly it differs from standard rice production practice in that paddy fields are not kept continuously flooded but their soil is just kept moist, with periods of drying, so that the soil's status is mostly aerobic rather than kept always saturated. As an innovation, SRI is 'not yet finished,' still evolving and changing, with further elaborations or variations coming from farmer practice as well as research evaluations.

The productivity gains that result from SRI changes in the management of plants, soil, water and nutrients, including greater water productivity and decreased water requirements, have now been demonstrated in some 50 countries by 4-5 million farmers on some 5 million hectares. SRI is spreading through a diverse group of stakeholders who support resource-limited, small-scale rice farmers in raising their output and incomes by using locally-available resources as productively as possible.

In recent years, the merits of SRI management as compared with the predominant anaerobic-soil (flooded) rice production systems have become better understood, based on both scientific as well as empirical data. The SRI production strategy gets manifested usually with six changes in agronomic practice: (i) the use of very young seedlings – about 8-12 days old – for transplanting, very quickly after removal from the nursery but also carefully and shallow; (ii) single transplant per hill, and (iii) wide spacing of transplants, from 20 x 20 cm to 50 x 50 cm, depending on variety and on soil fertility (higher yields come with wider spacing and lower plant populations when the soil is more fertile), so as to reduce plant populations by 80-90%; (iv) mainly moist (not saturated and flooded) soil water regimes, maintained through intermittent irrigation or small daily applications; (v) regular weeding through a rotary hoe that facilitates soil aeration as well as weed removal; and (vi) liberal use of organic fertilizers. These ensemble of practices was first assembled some 30 years ago by Henri de Laulanié, a Jesuit priest, who recognized that small rice farmers in Madagascar simply lacked the resources to invest in intensifying their rice cultivation practices through the recommended 'modern' techno[]logical package based on costly (and unavailable) external inputs and inadequate or non-existent extension support.

Relying on 'improved' varieties, the backbone of 'modern' rice production and indeed of industrialized agriculture in general, was not a promising approach for raising smallholder rice production in Madagascar. By manipulating the other crop management factors, including their interactions, Laulanié achieved large decreases in water requirements and spectacular yield increases using local varieties. There were water savings of 25-50% as well as greatly increased water productivity (crop per drop), by 50 to 100% or more. In essence, SRI crop management and water management represent 'integrated' agronomy. Through integrated management of the various crop-soil-soil biota-water-nutrient-space-time components, SRI seeks to capitalize on a number of basic agronomic principles that optimize the above- as well as below-ground plant growth and development, and the performance of the crop as a whole.

Thus SRI offers an opportunity to reduce water demand while enhancing yields, and also water productivity. It gives farmers an incentive to economize on water use. As has been shown in several studies, the most evident phenotypic difference with SRI is in plant root growth as well as in number of tillers and panicle size. Direct measurements confirm that SRI methods induce both greater and deeper root growth, which contributes to increased water use efficiency, and more nutrient uptake throughout the crop cycle. SRI rice roots as well as canopies resist senescence longer, compared with the shallower rooting and shorter duration of root functioning under continuous flooding. Rice plants grown with SRI methods take up more macronutrients than roots of conventionally-managed plants and give more grain yield per unit of macronutrient (N, P, K) taken up because SRI phenotypes function differently from rice plants grown under crowded, flooded conditions (Barison and Uphoff, 2011).

Evidence is accumulating that making the changes in the rice-growing practices that constitute SRI can result in win-win outcomes – for farmers, consumers, and the environment – in terms of water productivity as well as water savings. These gains are possible across a wide range of agroecosystems as noted below, and they are not limited to smallholders (Sharif, 2011). Nor are they limited to rice, as adaptations of SRI ideas and practices are demonstrating beneficial results with wheat, finger millet, sugarcane, teff, as well as a number of legumes and vegetables as the alternative plant, soil, water and nutrient management practices evoke more productive phenotypes for a variety of crop species (http://sri.ciifad.cornell.edu/aboutsri/othercrops/index.html). Although the greatest benefits come from using the full set of practices, and using them as recommended, there are demonstrable advantages from ‘partial SRI’. Based on the results of large-scale factorial trials in Asia and Africa, one can predict that in most of the cases reported, there are opportunities to achieve still-greater benefits from better use of SRI methods.

These methods, with appropriate adaptations, are effective in a wide variety of environments: tropical humid ecology (Panama), semi-arid regions on the edge of the desert (Mali), mid-altitude sub-humid tropical environment (Madagascar), sandy-marshy regions (southern Iraq), various dry and humid environments in Asia (India, Pakistan, Indonesia), even mountainous areas with a short growing season (northern Afghanistan). In each of these environments, farmers have found it possible through their modifications of standard rice-growing practices, according to SRI principles, to create microenvironments above- and below-ground that are favorable to more beneficial expression of rice plants' genetic potentials. A crop management and water management strategy such as SRI is *not an alternative* to getting and using the genotypes best suited to a given production situation; rather, it is a way to make the most of any given variety’s production capability.

SRI success is not dependent on farmers' using particular rice cultivars, although the highest SRI yields have come from combining its practices with high-yielding varieties or hybrids. Plant breeding has been, and will continue to be, important for in improving yield and other crop potentials. However, SRI methods can also raise the yields of most indigenous varieties, and where these command a higher market price because of consumer preferences, farmers may find these ‘unimproved’ varieties more profitable. This can help with the conservation of rice biodiversity.

Another important consideration is that SRI phenotypes are widely reported by farmers and observers to be less susceptible to pest and disease damage. In 2005-06, a systematic evaluation was done by the National IPM Program in Vietnam in eight provinces, comparing SRI plots with neighbouring farmer-practice plots. This found the prevalence of major rice diseases and pests (sheath blight, leaf blight, small leaf-folder and brown planthopper) to be 55% less on SRI plants in the spring season and 70% less in summer (National IPM Program, 2007). Farmers frequently say that with SRI management, their rice plants are resistant enough to crop damage that agrochemical protection is unnecessary or gives them no net economic benefit.

The SRI approach is an example of a paradigm shift, to more biologically-driven, agro-ecological strategies for crop production, in contrast to chemically-dependent ones. SRI experience is showing that better optimizing management of plants' environments for growth can achieve fuller expression of these potentials while minimizing overall water requirement and maximizing crop water productivity.

Such management can also help to buffer the effects of climate change (Uphoff, 2011). Further, especially if combined with Conservation Agriculture, it has the benefit of increasing long-term soil fertility. Inorganic fertilizers can be used with the other SRI practices, but organic fertilization is recommended and rewarded with higher yields. Soil organic matter is also enhanced by the larger and more longer-lived root systems which contribute much more root exudation and then through root residues that enhance soil structure and functioning.

6.4.4 Conservation agriculture in olive groves

The olive tree is a traditional crop in the Mediterranean region, which despite being perennial, barely covers the ground due to its broad framework of planting. Andalusia is a southern region in Spain, where around 60% of national production is obtained. There olive groves are commonly found in areas of steep hills as 75.8% of adult trees are planted in a range from 8 to 16% slope (Consejería de Agricultura y Pesca de la Junta de Andalucía, 2002). The introduction of mechanization in the mid-50s intensified the use of ploughs in Spain. Olive production involves a high risk of erosion due to the conventional soil management based on tilling. Indeed, intensive tillage is the common soil management system in olive groves (Pastor, 2004). Conventional agriculture drives to serious soil losses. Laguna and Giráldez (1990) estimated annual average losses between 60 and 105 t ha^{-1} $year^{-1}$. These losses greatly exceed the natural rate for soil formation, as seems to prove the progressive deterioration of vegetation in eroded surfaces. More recently, Vanwalleghem *et al*. (2011) quantified the effect of historical soil management on soil erosion over a 250-year period in olive groves. Mean rates of soil loss ranged between 29 and 47 t ha^{-1} $year^{-1}$. Water erosion was found to be 3 times more damaging than the one caused by wind.

Conservation agriculture (CA) improves olive groves environment. Although many measures have been planned, CA, understood as permanent cover crops (CC) in-between tree rows, is the best cost-effective system. Certainly, both for its simplicity and the degree of protection provided, which improves soil' structure and increases water infiltration. That extra water input, benefits the water balance. The management of soil with CC is proposed as a sustainable way to protect contamination of surface waters by trawling herbicide residues (Hermosín *et al*. 2011).

The use of CC reduces the power of the raindrops when reaching the soil surface. For the same rain, water loss in the plowing system is superior to that produced in the cover system (Espejo *et al*., 2007; Pastor *et al*., 2001). Erosion is no longer a problem with CA, as it reduces the output of organic matter adsorbed to sediment by decreasing water erosion and runoff (Gómez *et al*. 2009; Francia *et al*., 2006; Ordóñez *et al*., 2007). Gomez *et al*. (2011) studied runoff, sediment and nutrient loss from six sites in France, Spain and Portugal during 3-4 year. The study was focused in vineyards and olive groves and report CA reductions in erosion up to 97.4% when compared to conventional tillage. Moreover, authors report decreases in soil erosion, OM and associated nutrients in CC even when the quantity of runoff was not reduced when compared to conventional tillage.

CC enhances the surface flow dissipation, and maintains a network of roots at the center of the street creating biopores. These biopores improve the infiltration of water. Indeed, in a 4 year study in 9 plots, pores formed by roots and worms were found 25 times more on CA (Márquez *et al*., 2008).

Moreover, nutrient export decreases, maintaining a good quality of water that runs through the basin. Rodríguez-Lizana *et al*. (2007) studied in 3 fields, surface runoff, soluble P and Olsen P losses in sediment and concluded that CC reduced the total losses of both variables; between 7.6% and 36.5% the dissolved P loss, and between 16.3% and 56.4% that of the runoff. However, keeping the CC growing all the season may cause less yields in Mediterranean conditions. Water and nutrients could be a limit in the olive production (Berenjena, 1997). The delay in limiting CC growth, fairly common circumstance in Spain's countryside, causes a decrease in the moisture content. Even moisture could be the same in CC and tillage system if this delay happens. Márquez *et al*. (2007) report a relationship close to 1 (0.995), between volumetric moisture in the CC and tilled fields in a 4 season study in 9 fields located in South Spain, where climate conditions are drier than in the North. Therefore, it is mostly needed a proper CC management, for avoiding competition for water and nutrients among the olive tree and the CC in spring and summer.

6.5. Social and economic importance of water-related ecosystem services in farming systems

Managing the natural infrastructure in farmlands (soils and land cover) offers solutions to simultaneously meet three of the major natural resource challenges facing agriculture: water security, nutrient management and carbon storage. Together these provide a natural resource framework for sustainable agricultural intensification and are fundamental to the achievement of food security. The improved management of soil infrastructure and land cover also offers significant off-farm benefits through, for example, reduced erosion, improved water quality, climate change mitigation, and more efficient use of water in farming which reduces competition with other users downstream. The social and economic importance of natural water infrastructure in farmlands is, therefore, self-evident.

Examples of the on-farm benefits of sustaining or restoring natural infrastructure in farmlands are provided above. These benefits are an essential element if farmers are to have incentives to adopt natural infrastructure based approaches. Where on farm benefits are less obvious to, or immediate for, farmers, for example efforts to maintain off-site water quality, farmers may be encouraged to adopt more sustainable management through approaches such as payments for ecosystem services.

The scale of benefits on offer at the global and regional scales has not been quantitatively assessed in this chapter. However, previously provided figures suggest that most existing farmland is currently degraded and that 70% is moderately to highly degraded. The primary cause of this degradation is degradation of the natural infrastructure provided by soils and land-cover. Production on some degraded lands has been maintained through compensating with increased irrigated water and fertilizer use, and there is ample evidence that, overall, this approach is not sustainable. It is, therefore, not unrealistic to conclude that the social and economic benefit on offer is global food security.

Turner *et al*. (2004) considered approaches to the economic valuation of water for agriculture in the context of increasing competition for water and the need for better informed decisions regarding allocations. They proposed a number of generic principles that together form a powerful and comprehensive case for the wider adoption of a decision support system based around economic analysis, and which provides a thorough and powerful analysis of key issues related to agricultural use of water: economic efficiency and cost-benefit analysis; integrated analysis; an extended spatial and temporal perspective; functional diversity maintenance; long term planning and precaution; and the principle of inclusion. Furthermore they highlight that a transparent appraisal of water related projects, programmes or courses of action requires a comprehensive assessment of water resources and supporting ecosystems.

6.6 Carbon and water cycle interactions

Soil plays a key role in Earth's global carbon and water cycle. World soils collectively comprise the third largest global C pools (2000 Pg of organic C and 750 Pg of inorganic C to a depth of one metre). This total is 3.2 times the atmospheric pool (720 Pg) and 4.1 times the biotic pool (560 Pg). About 66 to 90 Pg C of the soil organic carbon (SOC) pool has been lost to the atmosphere due to conversion of native forest and grasslands systems to agricultural systems and soil cultivation (Lal, 2001). Most agricultural soils have lost 25 to 75% of their original C pool, and severely degraded soils have lost 70 to 90% of the antecedent pool (Bockheim and Gennadiyev, 2010), mainly due to mechanical disturbance through tillage which accelerates organic matter decomposition (Reicosky, 2004, 2011), and not returning organic matter not being returned to the soil from crops (residues, cover crops, manure etc).

6.6.1 Carbon and water cycles working together

The global energy, water, and carbon balances are strongly linked (Sellerset *et al*., 1997). Respiration is the largest C flux, contributing 120 Pg C/year to the atmosphere, with about half of the respiration losses being from heterotrophic soil respiration (Schlesinger, 1997). Plant photosynthesis is the key link between the energy (radiation) and carbon balances and evapotranspiration is the key link between the water and energy balances. Consequently, there are interactions between carbon and water with the soil being the mediating substrate. Also, the build-up of carbon in the soil is a dimension of 'natural infrastructure,' enabling soil systems to be better absorbers/users of water from rainfall and irrigation.

To improve the water balance for crop production of a given site it is possible to enhance water use efficiency as described in Section 3, or to increase the effective available water capacity of the soil. The latter can be achieved either by adding soil or by opening up the subsoil to increase its rooting volume and depth, by changing the soil texture, and by increasing the soil's organic matter content. Of all these interventions, only the break-up of hard pans or other rooting-depth-restricting structural soil layers, and the build-up of organic matter in the soil seem economically feasible and realistic measures to improve available water capacity.

A strong relationship has been found between the SOC pool, on the one hand, and the plant available water capacity (AWC) and the ability of soils to withstand drought, on the other (Hudson, 1994; Emerson, 1995; Gupta and Larson, 1979). In a literature review Hudson (1994) found highly significant positive correlations between SOM content and AWC for sand (r2 = 0.79***), silt loam (r2 = 0.58***) and silty clay loam (r2 = 0.76***) texture groups. In all the texture groups, as SOM content increased from 0.5 to 3%, AWC of the soil more than doubled. In his review, Hudson concludes that going from 1 to 6% OM by weight was equivalent to approximately 5 to 25% by volume. In general, the soil's available moisture content increases by 1 to 10 g for every 1 g increase in soil organic matter (SOM) content (Emerson, 1995). Thus, an increase from 1 to 2% of SOC in a topsoil (0-30cm) with an average bulk density of 1.5, would correspond to an additional AWC of 4.5 to 45 mm.

The improvement of soil structure through a more favourable particle aggregation and the consequent changes in soil porosity towards medium-sized pores capable of retaining plant available water, explain the increase of the effective AWC with the enhancement of SOC (Table 6.1). Although the higher SOM content in the top layer under the no-till treatment did not correspond to a higher AWC in this layer, there was a clear positive relation between SOM content and AWC in the monitored soil layer (0-30cm), albeit the additional effect of soil disturbance under the different soil tillage systems.

In addition, improved plant AWC leads to further positive feedbacks such as an increased ability of soil to withstand drought and to sustain biomass production, and also an increased water use efficiency (Lal, 2006). Both contribute to higher carbon inputs into the plant-soil system, thus enhancing SOC levels. This closes the loop and provides the necessary synergy to boost crop productivity and enhance the natural infrastructure.

There exists a strong relationship between agronomic production and the SOC pool, especially in low-input agriculture (Lal, 2010). Without regard for this knowledge, current agricultural land use continues to contribute to the decline of the SOC pool in vast regions of intensive crop production. It is estimated that between 66 and 90 Pg C of the world's total soil SOC pool has been lost to the atmosphere due to conversion of native forest and grasslands systems to agricultural systems and soil cultivation (Lal, 2001). Most agricultural soils have lost 25 to 75% of their original C pool, and severely degraded soils have lost 70 to 90% of the antecedent pool (Bockheim and Gennadiyev, 2010). There are numerous reports on the adverse effect of extractive, tillage-based agricultural land use on SOC concentration of arable land, mainly in tropical regions (Cole *et al*., 1993, Swift *et al*., 1994). Figures 6.4 and 6.5 and Table 6.4 make clear the enormous decline of SOC under different agro-ecological conditions and farming practices, also in temperate climates.

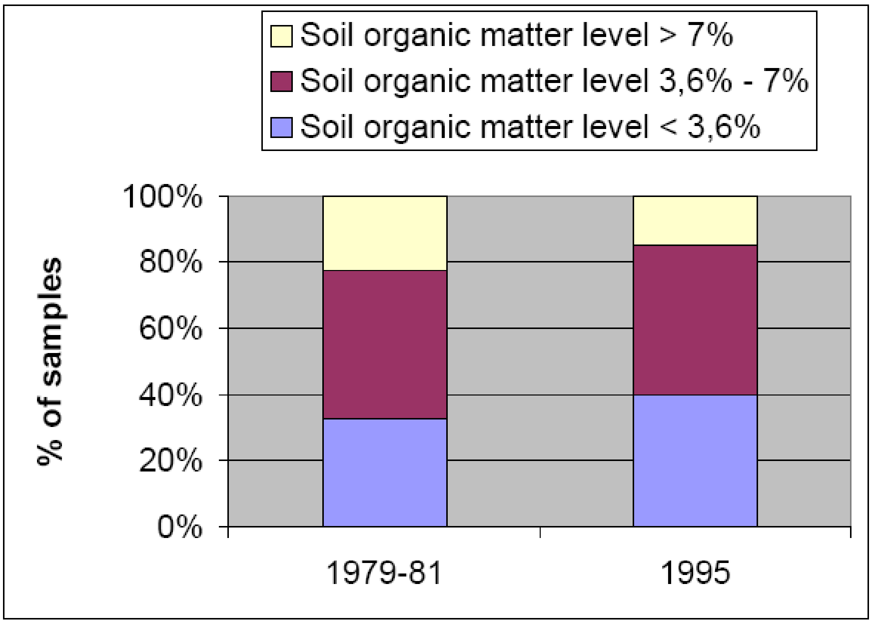


Figure 6.4: Organic matter content of agricultural topsoils: United Kingdom (England and Wales), 1979-81 and 1995, (Source: MAFF, 2000).

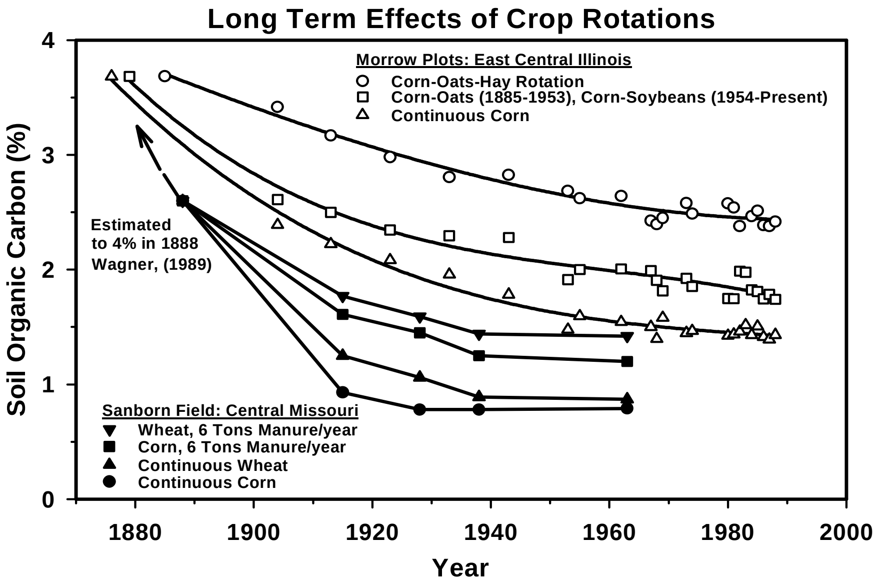


Figure 6.5: Long term effects of tillage and crop rotations on the evolution on soil organic carbon concentration (adapted from Odel et al., 1984, Wagner 1989, Reicosky, 1997)

Table 6.4: Effects of long-term cultivation on total organic C (g kg-1) in A-horizons in 8 different virgin (V) and cultivated (C) soils in Saskatchewan, Canada (adapted from Schnitzer et al., 2006).

Soil no.	Years of Cultivation	Total organic C (g kg^{-1})	
		V	C
1	50+	35.4	14.9
2	31	45.7	24.1
3	53+	54.4	27.6
4	85	39.6	20.5
5	48+	49.9	18.0
6	94	41.4	15.4
7	50+	54.4	27.9
8	53+	46.9	35.2
Mean		46.8	22.9

As outlined in Section 6.3, farming practices are capable to directly improve the water balance and the share of green-water available for biomass production. The improvement of the physical environment promoting infiltration, water retention, and reduced evaporation are undoubtedly some of the key effects of soil water-conserving farming practices such as minimum soil disturbance and permanent soil cover. Besides these interventions upon the soil physical environment aiming to improve soil water balance, the impact of agricultural practices and farming systems on soil organic carbon (SOC) concentration and pool is crucial for the water cycle and the crop-available water (Hudson, 1994).

The effects of any change in the cropping system or farming practices on soil physical and chemical changes are complex, and cause and effect can often not be clearly identified. Especially when it comes to evaluating the impact of changes in SOC on crop performance, it is difficult to tell, at least under field conditions, whether changes in nutrient availability, soil structure and consequent rooting conditions, soil water availability, etc., or the combination of two or several factors, were decisive for having differences in productivity.

For example, the wheat yield response to different levels of fertilization (Figure 6.6) on the same soil type with different levels of soil organic matter, obtained through long-term differentiated soil management (no-till + retention of residues vs. conventional tillage and residue removal), would be interpreted, in the first place, as the direct effect of an improved nutrient availability. However, other factors than nutrient availability certainly interacted and contributed to the higher yields at lower fertilizer inputs. Other favourable soil properties that are positively correlated to SOC content probably contributed to the enhanced crop performance, especially soil moisture availability under the prevailing water scarcity under Mediterranean conditions. Nonetheless, at any level of mineral fertilizer input, water use efficiency (same amount of rainfall) was considerably higher on the field with greater SOC.

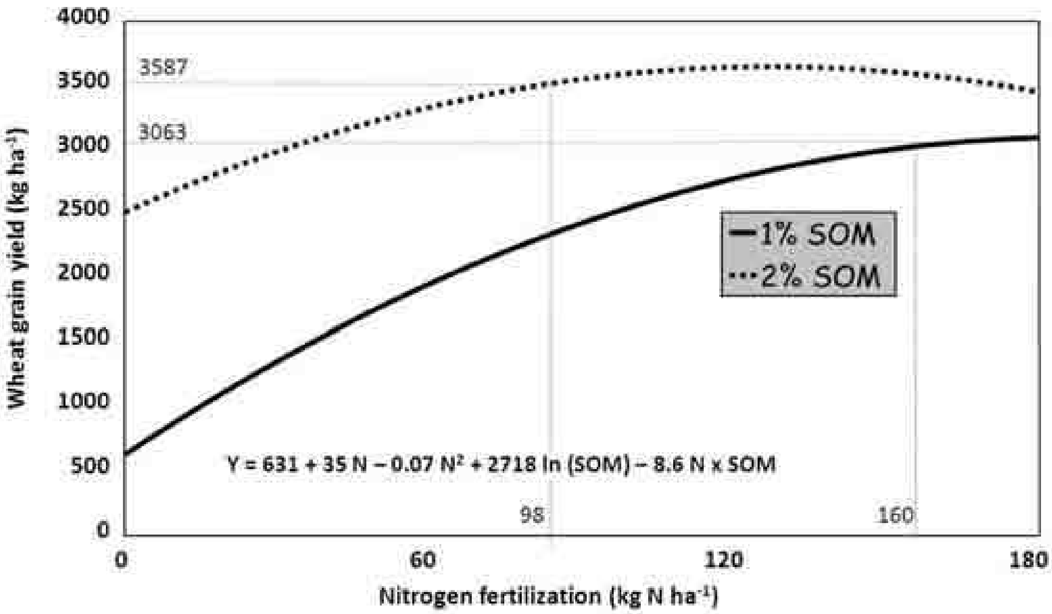


Figure 6.6: Wheat grain yield response to N-fertilization under different levels of SOC under water limited Mediterranean conditions (adapted from Carvalho et al., 2010)

In studies on the effect of long-term fertilizer management on water retention curves of soil aggregates, Liu *et al*. (2011) found the effects of using only chemical fertilizer to be limited when compared to the combined application of crop residue or manure and chemical fertilizer that significantly improved aggregate water-retention capacities.

Another positive, albeit indirect effect of higher total organic carbon (TOC) contents on soil water availability is the reduced soil compactability (Díaz-Zorita and Grosso, 2000). These authors found that the maintenance of high TOC levels are crucial to counteract compactability, i.e., to avoid the reduction of soil porosity.

A very clear and direct relationship between farming practices, SOC content, and AWC was reported by Fernández-Ugalde *et al*. (2009), who studied soil physical properties and crop performance on degradation-prone, semi-arid soils after 7 years of implementing no-till practice in comparison to conventional tillage (Table 6.5). Despite the very small difference in SOC between no-till and conventional tillage in the soil layer at 5 to 15 cm, the authors attribute the increased total water availability to improved structure characteristics and SOC content. This increased AWC doubled barley production under no-tillage in 2008, which was a very dry year.

Table 6.5: Soil organic carbon (SOC) and plant available water content (AWC) under no-tillage (NT) and conventional tillage (CT). Mean± standard error (adapted from Fernández-Ugalde et al. (2009).

	Soil depth (cm)		
	0-5	5-15	15-30
SOC (Mg ha^{-1})			
NT	11.15 ± 0.19*	15.95 ± 0.19	23.51 ± 0.39*
CT	8.06 ± 0.11	15.84 ± 0.23	20.82 ± 0.21
AWC (mm) [-33 to - 1500 kPa]			
NT	11.68 ± 0.20*	18.14 ± 0.40*	26.65 ± 0.55*
CT	7.87 ± 0.21	14.83 ± 0.21	20.89 ± 0.20

Values in the same column followed by asterisk (*) are significantly different at P <0.05 according to ANOVA.

Apart from the local scale benefits of SOC enrichment on improved water use efficiency, there are regional and global dimensions of carbon and water interactions, namely, with regard to climate change (CC) adaptation and mitigation. Most scenarios of the consequences of climate change involve changes concerning the total amount and variability of rainfall, including extreme weather conditions. In his review, Huntington (2006) forecasts an intensification of the global water cycle, without being precise in his conclusions about the consequences. However, from the examples above, we can conclude that soils with adequate levels of SOC are much better able to adapt to the adversities of both excess or scarcity of rainfall. Regarding the mitigation of climate change, there are numerous studies about the capacity of agricultural soils being an effective carbon sink (refs), thus reducing the global CC forcing potential of CO_2 emissions.

On the other hand, regional or global changes in soil water-holding capacity through various changes in farming practices and soil properties are also considered a very important issue in the modelling of global change scenarios as they would affect considerably the hydrological cycle through their impact on evapotranspiration (Ducharne, 2000).

According to Lal (2004), due to land misuse and soil mismanagement most agro-ecosystems contain lower SOC pools than their natural counterparts. This differential causes the low eco-efficiency of many agro-ecosystems whether managed on a traditional subsistence basis or commercially operated under rainfed or irrigated conditions in both developed and developing countries. Water-saving technologies in irrigated agriculture such as growing aerobic rice (Bouman *et al*., 2007) or using the System of Rice Intensification (SRI) methods (Uphoff *et al*., 2011; Kassam *et al*., 2011), as well as the adaptation of agriculture to climate change (Howden *et al*., 2007; Batisti and Naylor, 2009) through restoration of soil quality by improving the quantity and quality of the SOC pool (Lal, 2004) are, therefore, the big challenges for the adequate management of the main natural resources, soil and water.

6.6.2 Carbon offset scheme with no-till soil and water management

The greenhouse gas offset scheme operated since 2007 by the province of Alberta, Canada, is a good example of climate change mitigation through agricultural carbon offsets based on carbon sequestration, and integrated with water management as defined in a no-till farming system protocol which the participating farmers must follow.

This scheme allows regulated companies to offset their emissions by purchasing verified tonnes from a range of approved sources including agriculture projects (Haugen-Kozyra and Goddard, 2009). This compliance system for large emitters has provided a rich opportunity for learning on behalf of all players – the regulated companies, government, scientists, consultants, aggregator companies and farmers. Climate change legislation was amended in 2007 to require regulated companies to reduce their emissions to a set target below their 2003-05 baseline. If they could not achieve their target in any year, they could settle their accounts with any of three options: pay into a research fund at a fixed rate of C$15 per tonne CO_2e; trade emission performance credits if they were generated by any company reducing emissions beyond its target, or purchase verified offsets generated within Alberta using Alberta government-approved protocols. The latter option triggered interest and activities in developing protocols across all industrial sectors including agriculture. Offset tonnes trade at a discount to the C$15 fund payment option in order to cover the aggregation and transaction costs.

The Alberta government provides the enabling legislation and regulations. It also provide oversight of protocol development and approvals. Beyond that, the private sector invests in development of protocols, aggregation of offsets, and assembly of projects, third-party verification of projects, and the bilateral sales to the regulated emitters. A non-government agency, Climate Change Central, also plays a role of facilitator and is the designated operator of the Registry of the offsets. All verified tonnes are serialized and tracked by the registry through to the retirement (used for a compliance year) of a particular tonne.

The regulator/government ministry holds annual review meetings with the players in the market to review performance, new developments, regulatory changes, and guidance. The amount of offsets used by companies for compliance has been relatively consistent at about 36% of the total annual accounts (CCC, 2011). Agricultural offsets have contributed about 36 to 40% of all offsets. The most popular protocol has been the Tillage System protocol which acknowledges the soil carbon sequestration achieved through implementation of No-Till practices. The Tillage System protocol has contributed over 8 million tonnes of offsets worth C$100 million over the last five years of the offset system.

The offset system has had many co-benefits beyond reducing greenhouse gas emissions and reducing the C-footprint of industries. Scientists come together in helping to develop protocols and share a systems view of the production system under review. Science and policy come together and integrate to form protocols and develop a market. The private sector has a parallel function with aggregator and verification companies integrating their efforts and developing streamlined systems to bring offsets to market efficiently. Farmers have developed improved production and record systems. Very often the financial benefits to the farmer from adopting a particular protocol far exceed any offset payment for the greenhouse gas savings portion. All players are now further along the capacity curve to be in a better position to see and take advantage of other ecosystem goods and services opportunities.

6.7 Climate change

Climate change is a cross-cutting topic with regards to this work. The key point is that the primary impacts of climate change on ecosystems, and subsequently on people, are expressed through changes in hydrology (water availability) (IPCC 2008). The major impacts of climate change on food production therefore arise through changes in the mean availability of water for farming and, importantly, changes in the extent and frequency of the extremes of droughts and floods. Although there are regional and local differences, essentially, climate change increases risks and thereby reduces water security for farming. Adapting farming systems to climate change therefore centres on increasing the resilience of farming to water related risks. As noted above, farming currently already suffers from increasing water scarcity and risks. Climate change, therefore, is not considered a separate variable, but one that makes the existing situation and scenarios much worse.

The above mentioned situation with the loss of carbon storage in farmlands, and potential for carbon storage restoration within them, is also a major issue, and opportunity, regarding climate change mitigation. Mutually reinforcing linkages between the carbon and water cycle also clearly demonstrate how climate change adaptation and mitigation can be pursued together.

Because of the importance of the biodiversity-water-carbon cycle relationship, and how it underpins water security, nutrient cycling and food security in farmlands, "natural water infrastructure" approaches should be the cornerstone "climate smart agriculture" (www.climatesmartagriculture.org/) including being at the forefront of the strive to improve resilience of farming systems in the face of climate change.

6.8 Trends in global and regional attention to "natural infrastructure solutions" for agriculture

Trends in uptake of Conservation Agriculture, and related approaches, are mentioned above.
Rockstrom *et al*. (2009) proposed a new approach to global sustainability in which they identified planetary boundaries (with parameters) within which they expect that humanity can operate safely: climate change; biogeochemical nitrogen and phosphorous cycles; freshwater use; land system change; rate of biodiversity loss; ocean acidification; and stratospheric ozone. Sustaining and restoring water-related and dependent ecosystem services within agricultural landscapes is a major means to move towards safely operating within the first five boundaries simultaneously (climate, nutrients, water, land, biodiversity), and arguably, due to CO_2 reductions under achieved climate boundaries, also to the sixth (ocean acidification).
Because of the importance of food security, and the impact of farming beyond farmlands, agriculture related issues are high on the national and international policy agenda. The three primary global natural resources issues - food, energy and water security - certainly dominate much of the current natural resources related development debate. There is also increasing attention to the interrelationships between the three topics in dialogues such as the "food, water and energy nexus" (e.g. Hoff 2011).
There has been a significant discernible shift in policies and approaches in recent years away from managing agriculture as a sector which utilises external inputs (land, water, chemicals) towards viewing agriculture within a broader ecosystem and landscape setting. In particular, there is increasing attention to the role of ecosystem services in supporting agriculture: for example, paragraph 111 of the outcomes of the U.N. Conference on Sustainable Development 2012 ("Rio + 20") states "We also recognize the need to maintain natural ecological processes that support food production systems."
An emerging consensus on the need to sustainably intensify agricultural production, largely because the alternative is no longer a viable option and has significant implications for increased biodiversity loss, has prompted a shift towards better ecosystem management as a logical approach. For example, "*Save and Grow*" is a cornerstone of FAO's approach to food production and represents a paradigm shift towards the sustainable intensification of smallholder crop production (www.fao.org/ag/save-and-grow/). Some relevant elements of the strategy include:

- (i) Crop production intensification will be built on farming systems that offer a range of productivity, socio-economic and environmental benefits to producers and to society at large;
- (ii) Agriculture must, literally, return to its roots by rediscovering the importance of healthy soil, drawing on natural sources of plant nutrition, and using mineral fertilizer wisely; and
- (iii) Sustainable intensification requires smarter, precision technologies for irrigation and farming practices that use ecosystem approaches to conserve water.

Biodiversity is at the heart of such approaches and in particular with regards to how it underpins ecosystem services required by and influenced by farming.
The Comprehensive Assessment of Water Management in Agriculture (CA 2007) made a significant contribution to a strengthened science base, and increased awareness, of the challenges and solutions in the water-food nexus. Whilst recognising the opportunities to improve irrigation efficiency, importantly the assessment concluded that the greatest potential increases in yields are in rainfed areas: but with the caveat that only if leaders decide to do so will better water and land management in these areas reduce poverty and increase productivity. Given the limited opportunities for managing rainfall directly (although managing evapo-transpiration from land cover is an exception – see chapter 2), sustaining the natural water infrastructure within agricultural landscapes is a large, if not the main, sustainable strategy for improving yields in rain-fed areas.

Sustainable management of the natural resource base supporting agriculture is one of the three major strategic objectives of the Consultative Group on International Agriculture Research (CGIAR). The CGIAR Research Programme on Water, Land and Ecosystems (http://www.iwmi.cgiar.org/CRP5/) combines the resources of 14 CGIAR centres and numerous external partners to provide an integrated approach to natural resource management (NRM) research, and to the delivery of its outputs. The programme focuses on the three critical issues of water scarcity, land degradation and ecosystem services, as well as the CGIAR System Level Outcome of sustainable natural resource management. It will also make substantial contributions to the System Level Outcomes on food security, poverty alleviation and, to a minor extent, health and nutrition. The current topic is effectively already embedded as major component of this research approach.

At the level of international agreements, the CBD Strategic Plan for Biodiversity and the Aichi Biodiversity Targets represents a strong framework for action in this subject area. Some of the important links between the Aichi Biodiversity Targets, food, water security and development goals were illustrated in document UNEP/CBD/SBSTTA/INF/19 (http://www.cbd.int/doc/?meeting=sbstta-15). Better management of the role of biodiversity in agricultural landscapes, that is, soil biodiversity and land cover, are central to the achievement of Aichi Biodiversity Targets 4, 7, 8 and 14 (among others). In addition, restoring water-related land functionality would be a major component of efforts towards ecosystem restoration (Article 8f of the Convention itself, and as to be considered at COP-11 re. document UNEP/CBD/COP/11/21). In fact, much current ecosystem restoration is probably already driven largely by interests in improving water security. The topic is also central to the objectives and strategies of the United Nations Convention to Combat Desertification (UNCCD). Loss of soil moisture, by definition, is the direct driver of desertification and sustaining or restoring soil moisture, therefore, the primary means to combat desertification. For example, this topic has a high profile in considerations of indicators for the impact of the strategic objectives of that convention (UNCCD 2011).

All of the above illustrate increasing recognition of the importance of ecosystem services with regards to sustainable water and land management and agriculture. Different interest groups commonly use different terminology. But for present purposes, all of the above examples centre on sustaining and/or restoring the functionality of natural water infrastructure in agricultural landscapes.

The extent to which such recognition and approaches are mainstreamed into policies in practice is, however, difficult to assess. For example, much of the dialogue on improving water storage and supply for agriculture is still dominated by a focus on irrigation and in particular large infrastructure development approaches that dominated advances in improved agricultural productivity in the 1960s and 1970s. Although there are opportunities for improved irrigation to contribute, a better approach is to articulate the need as “water storage” (not irrigation) and thereafter to consider the most effective means to store water and make it available to plants. In this regard, there is a strong argument that the safest and most sustainable and efficient place to store water for agriculture (and indeed other purposes) is very often in the ground (soils and groundwater). The menace of water scarcity, worsened by climate change factors, carries the danger of driving policies to try to build more dams and reservoirs, to mechanical and engineering solutions, rather than biological and vegetative solutions; and to large-scale dams rather than small-scale, dispersed, distributed water within catchments. This chapter has attempted to contribute to generating some new thought and new perspectives, and maybe having some impact on government and investment decisions in these regards.

CHAPTER 7 Understanding how institutional arrangements support natural infrastructure to ensure water security

7.1 Introduction

Ecosystems provide services by maintaining water flows and supplies, regulating water quality and minising water-related disasters (Emerton and Bos, 2004). Rapidly expanding population and economic growth, particularly in developing countries, is requiring more water to meet growing food and energy demands. This is contributing to difficult trade-offs in how water is allocated which, in many instances, is resulting in limits to economic growth and/or substantial ecosystem degradation. However, it is not simply limited availability that presents economic and social difficulties, but rather the variability of water in space and time that presents the greatest challenge. This variability presents economic costs to both vulnerable people and national economies. Droughts and floods, can have huge economic consequences for countries and it is often the most vulnerable communities that bear the greatest impact. If the climate change predications are correct, these economic impacts are likely to increase in the future (Lenton and Muller 2009; WWAP 2012). Furthermore, although significant progress has been made, globally, there are significant and on-going challenges, particularly in developing countries in delivering sustainable water and sanitation services to communities. This is partly because many water and wastewater utilities are operating with aged infrastructure that requires large capital investments for upgrades or replacement. Affordability of infrastructure upgrades presents on-going challenges, particularly in the current economic climate.

Within this global context, the objective of this book/piece of work is to expand the traditional understanding of infrastructure, that is narrowly focused on the physical structures associated with providing and distributing water, to a broader understanding to include the role that natural ecosystems have in ensuring the reliability and resilience of water resources, as well as reducing vulnerability to water-induced disasters from floods and droughts. The services provided by ecosystems are often poorly understood and inadequately communicated, contributing to them receiving inadequate attention by decision-makers. Other chapters/contributions have addressed this by clarifying the available science and outlining knowledge gaps. However, improved science, while crucial, is only part of the solution. In order for this understanding to be integrated into decision-making, there is often a need to reform institutional arrangements. Natural infrastructure solutions often require wide-ranging solutions that expand beyond the jurisdiction of many water managers and usually require the engagement of a wide-range of stakeholders. The implementation of the solutions may require difficult trade-offs and changes in land-use practices which are not easily implemented without wide-scale political support. Natural infrastructure solutions cannot work everywhere but they can sometimes offer viable alternatives, or complementary actions, to traditional infrastructure. However, integrating these solutions requires an understanding beyond the science of ecosystem services to the institutional arrangements within in the water sector as well as the enabling mechanisms that support implementation. An understanding of these dimensions will assist decision-makers in understanding the context and the tools that are required to better integrate ecosystem services into decision-making.

7.2 Institutions

This first section of this chapter provides a definition of institutional arrangements in the water sector, as distinct from organisational structures. The second section outlines some of the reforms that have taken place in the institutional arrangements associated with water resource management, specifically those of IWRM as well as private sector involvement in water service delivery. These water reform contexts are then placed within the context of broader reforms in public sector institutions. In conclusion, three principles are outlined as important considerations for mainstreaming natural infrastructure: the importance of the broader socio-political context; the importance of working strategically within non-ideal arrangements and; the importance of developing appropriate ways to measure performance to match government priorities.

7.2.1 Definition of Institutional Arrangements

Institutions can be defined as an expression of the formal and informal rules and norms that shape the interactions of humans with each other and with the environment (Cortner 1998). Or more simply, they are the "humanly devised constraints that shape interaction" (North 1990). Consequently, institutions have widely diverse interpretation and they reflect "different disciplinary perspectives and theoretical traditions" (Saleth and Dinar 2005). However, within the water sector, there is general understanding that institutional arrangements encompass the "rules that together describe action situations, delineate action sets, provide incentives and determine outcomes both in individual and collective decisions related to water development, allocation, use and management" (Saleth and Dinar 2005). Institutions can be broken down into the institutional environment and institutional arrangements. The environment includes the fundamental political, social, and legal rules that guide production, exchange, and distribution. Institutional arrangements provide structure within which members of a society cooperate or compete, including legal, policy and organisational components (Saleth and Dinar 2005) as per Figure 7.1.

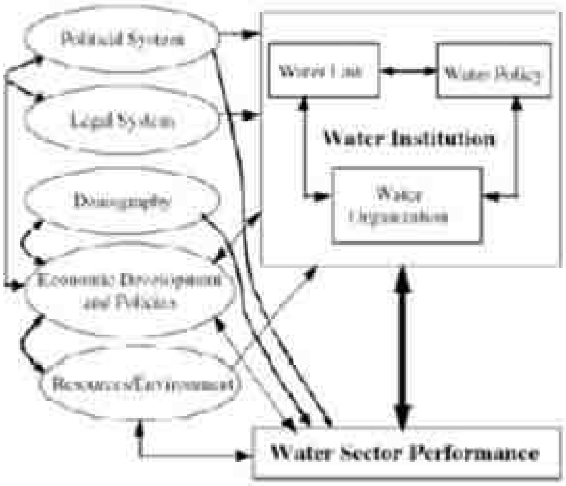


Figure 7.1 Water institutional environment: a partial representation. Source: saleth & Dinar (2004).

Within the water sector, particularly within developing countries, it is important to acknowledge that the formal policies, rules and regulations usually differ substantially from the application in practice. Understanding institutional arrangements requires an awareness of and a distinction between "the rules" and "the rules of use" (Ostrom 1992). The interpretation of formal sets will differ depending on the nature of the natural resource, water demands and socio-economic pressures.

7.2.2 Organisational arrangements

Institutions are therefore broader than just the organisations involved in decision-making. Organisations are a subset of institutions and are usually created within an existing web of institutions (Svendsen *et al.* 2005). However, a first step in understanding institutional arrangements is to identify the organisations involved. These are usually divided into those involved in water resource management (catchment planning, protection of the resources, allocation of water between sectors, infrastructure planning for large-scale water supply interventions) and those involved in water supply and waste services (distribution of water for domestic, municipal supply, waste water treatment). The actual departments involved in these functions will differ from country to country. Figure 7.2, provides an example of the organisations involved in the South African water sector. In terms of mainstreaming natural infrastructure, the number of organisations involved would likely involve much more than those with a direct water management or water supply mandate, and could include departments involved in agriculture, environmental authorisations, land use change, zoning and urban planning. Establishing integration across these departments presents a barrier for implementation.

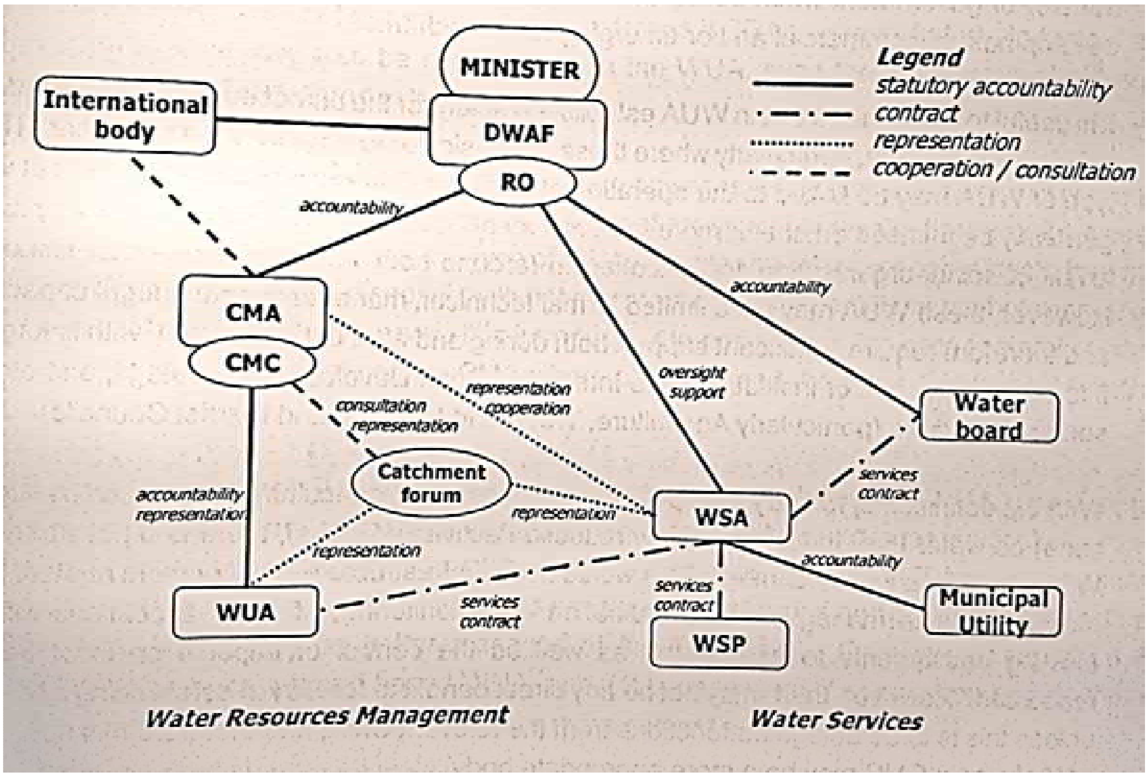


Figure 7.2. Primary institutional relationships between organisations within the South African water sector (Pegram and Mazibuko 2003) *Reproduce with acronym explanations*

7.2.3 Integration of institutions as a barrier to natural infrastructure

Institutional arrangements that support natural infrastructure require integration across multiple organisations. Establishing these arrangements has been found to be a key barrier to ecosystem based approaches. Brown *et al.* (2009) reviewed 53 studies drawing on local, national and international literature from the field of integrated urban water management and other similar fields to identify the main barrier to implementation. The review found that, while multiple barriers were identified, the most commonly identified barrier was the lack of a coordinated institutional framework. These findings have been supported by similar conclusions in non-urban environments (Svendsen 2005; Cortner *et al.* 1998). There is a need to shift from traditional, linear, 'old-world' approach to those that are more adaptive, participatory and integrated and which recognise the totality and integrated nature of the water cycle (Brown et al 2009).

Over the last few decades, the institutional arrangements governing water have undergone significant changes (Saleth 2006) some of which support more integrated approaches. This includes the enactment of new water laws, policy reforms and the establishment of new and restructured organisations. Many of these changes have been a product of purposeful reform rather than institutional evolution. A key reform, relevant to ecosystem services, is the shift towards the establishment of integrated approaches to water resource management, sometimes terms Integrated Water Resource Management (IWRM). While the concept is contested, the philosophy supports catchment-wide and integrated approaches. The following section provides an overview of how IWRM approaches emerged within the water sector. Understanding these institutional histories is important as institutions are path dependent, which means that history does matter (Saleth and Dinar 2004). The directions and future path of change within the sectors cannot be separated from past histories. Understanding path dependency is important in understanding future directions and the potential costs associated with change. Understanding the evolution of IWRM also provides a context for understanding why certain approaches to reform fail, which will be discussed in the final section.

7.3 Evolution of water resource management institutions

Tracing back to early civilizations, and throughout history, water has been closely association with the development of human societies. The beginning of irrigated agriculture traces back to 7000 BC, when ancient civilisations were drawn to the fertile soils and irrigation potential of riverbanks. The drainage basin formed the framework for the human settlement of early civilisations which guided the direction of primary settlement, river navigation and the context for irrigation works (Smith 1969). These civilisations also developed hydraulic engineering such as canal and storage systems for irrigation to a degree that has not been surpassed until modern times. In response, a single centralised political authority evolved to maintain the canals and control the distribution of water (Smith 1969). The local impacts of this water use may have been significant but, on a larger scale, river flow regimes were largely unaffected (Svendsen *et al.* 2005). However, as world populations grew, human demands for water expanded and urban concentration required increasing abstraction to meet agricultural and industrial demands. Technological advances of the time -- including concrete, steam and electric power dredges, dynamiting equipment and particularly hydroelectric power and long-distance transmission—resulted in the transportation of water over longer distances. Consequently, the 20th Century saw a period of remaking and manipulating natural hydrology to meet growing population demand for domestic supply, sanitation, food, fibre, energy and industrial needs (Svendsen *et al.* 2005). Expanding water supply was seen as the easiest and most cost-effective solution as water was relatively abundant and the environmental impacts were incremental and not immediately apparent to society.

However, as the impacts of water abstraction where manifest in the natural ecosystems, livelihoods were impacted, through declining fisheries and other natural products. As societies and their demand became more complex, so did the governance instruments that were required to manage them (Svendsen 2005). Consequently, administrations that were previously only responsible for the hard solutions of building infrastructure to supply water, where required to pay greater attention to the protection of ecosystems (Lenton and Muller 2009). This required the development of a new set of skills and approaches that were beyond the ambit of traditional supply-orientated water management institutions.

7.4 International influence on development of the river basin concept

In 1956, the UN Secretary General stated that river basin development is an essential feature of economic development and that integrated river basin development would promote human welfare (Saha and Barrow 1981; Teclaff 1991). Individual water projects would not be undertaken unless there were broad plans for the entire drainage basin (Teclaff 1996). The United Nations Water Conference, held in Mar del Plata in 1977, encouraged countries to consider “as a matter of urgency the establishment and strengthening of river basin authorities” (Recommendation 48d). The acceptance of catchment-based management was spurred on by the rising tide of environmental awareness, which began in the 1970’s and led to the emergence of the ecosystem concept as a guiding principle for resource management. Reynolds (1985) defined the ecosystem approach as:

> The anticipatory approach to planning of river basins and general problem solving that is based on the knowledge of the operation and interrelationships of systems in nature and, in consequence, the necessity of ecological behaviour and desirability of adoption of an ethic of respect for other systems of nature (p. 41).

The growing acceptance of the ecosystem concept led to a natural recognition of the river basin as the appropriate management unit. The management of water resources at this level allowed all activities, which affected the basin’s ecosystem, to be managed through a more holistic approach.

7.5 Evolution of Integrated Water Resource Management

The 1990’s, saw widespread evidence that much of the infrastructure constructed for economically beneficial purposes such as: flood control, hydropower production, and irrigation, had caused significant damage to freshwater ecosystems (Teclaff 1996). This initiated numerous projects that attempted to undo some of the ecological damage and to initiate a more integrated approach to water management. The experts who prepared proposals for the United Nations Conference on Environment and Development at Rio de Janeiro in 1992 specified that “integrated water resource management, including the integration of land-and water-related aspects, should be carried out at the level of the catchment basin or sub-basin level, taking into account existing inter-linkages between surface and ground waters” (at 519, paragraph 20).[1]

In 1992, the Dublin Statement on Water and Sustainable Development made an important contribution to supporting the principle that effective management of water resources demands a holistic approach that links social and economic development with the protection of natural ecosystems (Teclaff 1996). Provoked by these combining economic, environmental and social factors, the river basin became recognised as the most suitable organisational unit for water management.

7.6 Challenges to the concept of IWRM

IWRM therefore evolved out of these declarations as an alternative to the single sector, top-down approaches of the past. The concept is defined by the Global Water Partnership as a process that promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems (GWP 2000). Despite its initial acceptance, the concept has received much criticism, as Merry (2008) argues, “with all respect to the Global Water Partnership, Waternet, the World Bank, and most of the water management ‘establishment,’ it is time to abandon Integrated Water Resources Management (IWRM) as a guide for implementation.” Others, notably Biswas 2008, have argued that IWRM is simply unusable, or unimplementable, in operational terms. There are accusations that despite much intellectual debate, the implementation of IWRM has been minimal, even indiscernible in the field.

Lenton and Muller (2009) respond to this critique by reviewing case studies that reflect IWRM principles and approaches. The case studies find common elements to the approaches which recognise:

- The unitary nature of water resources;
- The physical interventions that could be adopted to manage water resources;
- The limits to these physical interventions; and
- The need for an institutional framework that bring stakeholders together equitably and gives voice to the weak as well as the powerful, that seeks to achieve a balance of interests that recognised the value of the water concerned, which identifies the environmental dimension of water management as a separate use or as a desirable outcome and which seeks to develop organisations that are able to promote this overall approach (Lenton and Muller, 2009).

If IWRM is seen as a fixed prescriptive toolbox, requiring implementation of all tools without consideration of context and sequencing, then IWRM is indeed an unhelpful approach (Lenton and Muller, 2009). But referring to case studies across scales and geographies, IWRM should not be viewed as a prescription but a framework or philosophy from which the problem of communities and nations can be addressed in ways that differ from the single-sector approaches of the past. IWRM cannot be seen as a blueprint for action but rather the operationalization of key principles that are implemented through approaches that are best suited to the political, physical and socio-economic context of the particular water environment. The connections across a basin are not simply hydrological and ecological but inherently socio-political. Critically, IWRM should not be seen as an outcome in itself but as an approach which requires on-going adaptation and review of interventions and approaches that respond to on-going water management challenges (Anderson et al 2008).

[1]See the protection of the Quality and Supply of Fresh Water Resources: Application of Integrated Approaches to the Development, Management and Use of Water Resources, U.N. Conference on Environment and Development, Agenda Item 21, Ch 10, 1 at 22 par. 19, U.N. Doc. A/Conf.151/PC/100/Add. Reprinted in 1 Agenda and the Unced Proceedings 513, 519.

7.7 IWRM and broader reforms

While IWRM approaches emerged mainly out of a rising tide of environmental awareness and dissatisfaction with the single-sector response, their development was also influenced by broader reforms within public administrations, specifically those of decentralization. Of particular interest is that at the same time as the concepts of IWRM were gaining momentum, major reforms in the public sector were occurring. The emergence of New Public Management (hereafter NPM) is defined as one of the most striking trends in public administration (Hood, 1991). The IWRM approach supports the NPM logic of decentralizing activities to smaller and more logical delivery units that have autonomy in delivering outcomes within a specified budget allocation (Lenton and Muller 2009).[2] These broader government-wide reforms are seldom discussed in the water sector literature. Broader governance arrangements are particularly relevant in the water sector because water is a deeply political commodity, consequently, “water management is inherently political and embedded in a larger institutional context” (Merrey 2008). Water management happens within the politically contested space of scare resource allocation, implemented by administrative authorities that may have divergent political agendas and --particularly in developing countries-- limited resources and capacity to effect change. Politics is often at the centre of the problem. As Muller and Lenton 2009, argue, “one of the challenges of promoting better water management is to do so in a manner which is compatible with broader approaches to governance and public management.” (p 4) The following section will describe the characteristics of the NPM reforms and how they had an influence on the delivery of water supply services.

7.8 New public Management and Reforms in Water supply Sector

Although NPM is loosely defined, it is characterised by a set of administrative doctrines that dominated the reform agenda of public administration from the 1970s onwards. Although the approaches began predominantly in the OECD countries (UK, Britain, Australia) they were also prominent across developed and developing countries in different forms. NPM emerged as a critique of inefficient bureaucratic processes and sought to improve the provision of public service delivery (Dzimbiri 2008). The approaches responded to the growing dissatisfaction with a large rigid bureaucracy that held excessive control over the economy and which was unable to delivery effective public services. NPM includes reforms such as:

> “increased autonomy for managers (letting mangers manage), accountability for results (“making managers manage”), entrepreneurial management and a variety of market mechanisms such as privatisation, contracting out, and competition” (Deleon 2005, p. 104).

From the 1980s, NPM-type reforms were geared towards “enhancing efficiency, productivity, improved service delivery and accountability” (Dzimbiri 2008). Hood (1995) argues that NPM brought in a new approach to public management, shifting the emphasis away from process, towards a greater emphasis on accountability for results. NPM brought a specific focus on trimming fat and “avoiding slack” (Hood 1995) and is underlined by values that emphasise frugality and efficiency, with an emphasis on “keeping things lean and purposeful” (Hood, 1995).

NPM also supports the ideal of liberating public managers to have more discretionary power in delivery outcomes, rather than following specific rules (Hood 1991). A key characteristic of the delivery mechanism was to unbundle services into decentralised units and to organise outcomes by service or product. While many argue that the approaches reached their peak in the 1990s and are now declining in emphasis (Andrews 2008), the impact of the reforms are still felt and the approaches bought in a new approach to public administration.

[2] See Oats (1968)for a discussion on Decentralisation and budget reforms

7.9 Implications of New Public Management to the water sector

The involvement of private actors in the water sector was one response to NPM reforms. In the 1990s, many governments embarked on ambitious reforms of the water supply and sanitation sector. Specifically, public private partnerships (PPP) emerged as a new approach to solving inefficient public utilities. Between 1990 and 2009, more than 260 contracts were awarded to private operators for the management of urban water and sanitation utilities in the developing world (Marin 2009). Many of these contracts were highly controversial because of community opposition and mixed results that were achieved. Consequently, much of the initial optimism concerning private sector involvement in the water supply and sanitation sector has now faded (Schwartz, 2008). Since 2001, there has been a decrease in the annual number of PPP contracts awarded in the water and sanitation sector and there is now a more cautious approach from all stakeholders towards PPP contracts, seeing them as only one institutional approach to water sector reform. Some argue (Marin 2009) that the debate against PPPs was driven more by ideology than objective results and that more performance evaluation is required to assess the performance record of private sector involvement. In 2009, only an estimated 7 per cent of the urban population in the developing world was served by private operators, and so the public sector remains the main provider of water supply services (Marin 2009).

However, despite the small number of people currently served by the private sector, the involvement of the private sector, contributed to a change in the management ethos of public water utilities. Many utilities were reformed to be more commercial, although they remained publically owned. In some instances, the reforms were introduced as precursors to full privatization (Dagdeviren, 2008) which was never realised. Institutional arrangements and management practices, usually associated with the 'private sector', were widely introduced (Swartz 2008). According to Schwartz (2008), the commercialisation reforms in the water supply and sanitation sector shared some the following characteristics:

1. Increasing the level of autonomy of the utility;
2. Separating regulatory tasks from service provision;
3. Creating quasi-competition in the water sector;
4. Increasing tariffs to cost recovering levels and increasing customer orientation; and
5. Increasing accountability for the results produced by the utility (p. 50)

Although these types of reforms were common globablly, those that were most successful were those types of arrangements that were adapted to socio-economic context of the countries involved (Anderson and Janssens 2011). The World Bank has refined its approach to privatization to acknowledge that one needs to work through markets and the state and that one model does not fit all, specific circumstances do matter (Fine and Bayliss 2008).

7.10 The way forward for mainstreaming natural infrastructure

Influenced by changes in the public sector at large, institutional arrangements in the water sector have faced significant reforms. An analysis of these reforms outlines three areas that can guide institutional arrangements to mainstream natural infrastructure.

7.10.1 Understand the context

A critique of both IWRM reforms and privatisation reforms is that the implementations of new approaches are often viewed as depoliticized process. Water management happens within the politically contested space of scare resource allocation, implemented by administrative authorities that may have divergent political agendas and --particularly in developing countries-- limited resources and capacity to effect change. Institutional arrangements are deeply embedded within governance framework and water management institutions and policies are frequently contested and the outcomes of political practices (Svendsen 2005). Past reforms in the sector have shown that imposing institutional models uncritically in vastly different socio-ecological settings seldom results in desirable outcomes (Shah *et al.* 2005). IWRM has been criticised because the blueprint models that

were successful in many developed countries were applied uncritically in developing countries that had vastly different conditions and challenges. Within the river basin, approaches need to be shaped to meet the hydrology of the basin, the demography and socio-economic conditions and the institutional arrangements (Shah *et al.* 2005).
Based on Nobel Laureate Oliver Williamson (1999), Shah suggests that there are four levels that can be used to understand how societies adapt their institutions to changing demands (Figure 7.3). Level 1 is defined as the social embeddedness level, consisting of customs, traditions, mores and religions. Institutions within this level take a long time to change because there is little deliberative, calculated choice. The second level considered the institutional environment which is presented through formal rules, constitutions, laws, property rights. This includes the rules of the game which can be changed through deliberate action but which will likely take time to reform. However, the rules of the game may be different from how the game is played. Level three is concerned with enforcement of contracts and property rights. While the final level, is concerned with aligning incentives. Shah argues that, particularly in developing countries, the water sector tends to give insufficient attention to the first level which results in new approaches being inconsistent with long standing norms and traditions. IWRM is not implementable when it is seen as a prescriptive list of actions to be implemented regardless of the context and embedded aspects such as tradition and customs Furthermore, there is insufficient attention paid to how these four layers work together. For example, perfect institutional arrangements will only make a difference on the ground if they are supported by realistic enforcement mechanisms and incentives. Particularly in developing countries, there is often insufficient attention paid to the practical difficulties associated with implementing ideal solutions that may have been formulated in countries with more capacity and resources to support implementation.

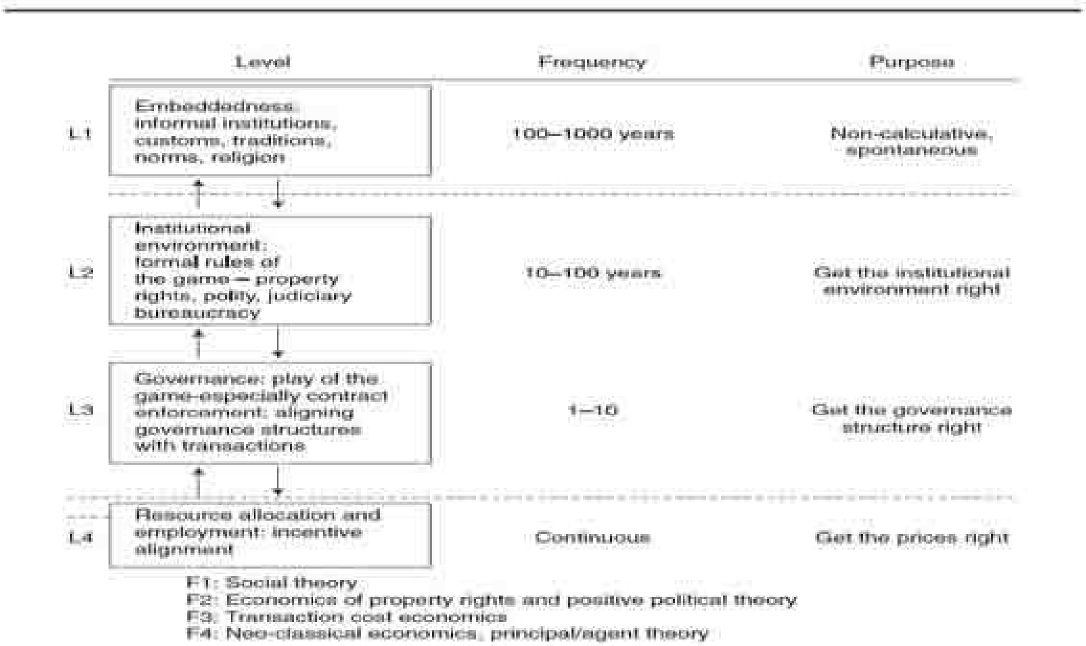


Figure 7.3: The four levels of how societies adapt their institutions to changing demands. From Shah et al 2005, based on Oliver Willamson (1999).
What precisely are the implications of the above for mainstreaming natural infrastructure solutions? Would one be to focus on levels L4 and L3 (in the short-term) which would suggest that promoting "cost" aspects on infrastructure (L4) ??? (which is an unsurpring conclusion) - and what exactly is required re. L3?

7.10.2 Good enough institutional arrangement

IWRM reforms have been criticised for requiring too much and for expecting too much integration and for being too prescriptive in their responses. Merrey (2007) articulates a way forward by proposing a more "problem-based" approach, which involves "focusing on specific issues and problems, prioritize them, identify options for solving the highest-priority problems, identify the constellation of interested parties, actors and institutions, and negotiate a way forward." He argues that IWRM only offers a systems context to understand the implications of certain solutions. This approach is consistent with others working across public sector reforms in developing countries. Referring to work in establishing good governance in developing countries, Grindle (2004) argues that the call for good governance as a precursor to poverty alleviation is unrealistic as good governance reforms touch on virtually all avenues of the public sector. An alternative is to focus on "good enough" governance which focuses on the sequencing of key strategic priorities in the short and long term. This approach accepts the need for a more nuanced approach to how institutions evolve and existing government capabilities. There is a need to be more explicit about trade-offs and priorities in a world where all good things cannot be pursued at once (Grindle 2004). In terms of mainstreaming natural infrastructure, there is a need to focus on those activities that can achieve results within a specific context. A good way to design a strategic approach is to recognise that no matter how ineffective an institution may appear from the outside, there are always processes or approaches that work better than others within that context (Grindle 2004). Starting with questions that seek to fully understand why some processes achieve results within a particular water context and build on them is likely to be a more successful approach than asking questions about how to reformulate long standing institutional arrangements according to a "formula" that may have worked in a completely different water management context.

In order to mainstream natural infrastructure into institutional arrangements in the water sector, a key recommendation is to strategically identify those mechanisms that are likely to be successful within a specific context and to sequence them realistically within the short and long term. As discussed in the previous section, some approaches, such as establishing basin -wide planning processes which alter land use practices, may only be successful with a complete reform of customary land use arrangement this may take decades to achieve. Other options, such as tweaking certain designs within existing traditional infrastructure to better service ecosystem protection, may be more easily achieved in the shorter term.

7.11 Performance management and measuring performance

The most interesting impact of NPM reforms is the emergence of the increased role of performance management. Across all government agencies, performance management processes and measures have been increasing (Meir and Hill 2005). Performance management requires target setting, benchmarking and monitoring outcomes. As central government services shift from implementation to contract management, the role of government is altered from "creating public value" (Moore 1995) to managing the performance and output of private sector contracts or the outputs generated by other public entities. Contract management and performance management have become of paramount importance to government administrators. Public sector managers may now have more creative freedom in achieving outcomes but they are held to a clear bottom line to reach specified outcomes. A critique of NPM is therefore that in an attempt to remove bureaucratic controls, government's role has shifted to that of auditing and contract management (Meier and Hill 2005). These pervasive reforms to public sector institutions have implication for the water sector. Firstly, as target setting becomes a goal of government, targets are often defined in terms of achievable, easily quantifiable outcomes that can be obtained over a short time period. Furthermore, targets that are defined within one government department are likely to be more easily defined with supporting data than targets that require cooperation and data gathering from multiple agencies. Defining specific, measurable outcomes of natural infrastructure approaches, compared with traditional infrastructure, is particularly challenging. New built infrastructure can be measured with often direct benefits, whereas the full benefits of alternative approaches are more difficult to communicate across multiple government service delivery agreements. Although the techniques associated with measuring the benefits of natural infrastructure are improving, the inter-disciplinary techniques are difficult to coordinate and data collecting to support the outputs is more time consuming for government administrators than single sector responses. An important way to mainstream natural infrastructure into public sector decision-making is to develop ways to clearly demonstrate how these approaches achieve specific government targets.

Secondly, as senior officials are given more freedom to deliver outcomes, they may be freed of certain type of administrative control. In some instances, particularly in countries with weaker fiscal management structures, the weakening of these controls can lead to increased corruption as government officials fall prey to “rent-seeking” as private companies seek lucrative contracts for public service delivery. Of the US$11.1 trillion the world is predicted to spend on energy infrastructure between 2005 and 2030, US$1.9 trillion is estimated to go toward hydropower (IEA 2006). These large amount of money can create opportunities for bribery and fraud and other forms of corrupt behaviour. Civil works contracts are often identified as the largest budget line, accounting on average for 60 per cent or more of total project, which leaves the dam sector vulnerable to corruption if insufficient controls are in place. Understanding who is benefitting from existing arrangements is an important gage to assess, realistically, what changes are possible. In some instances, real changes in how infrastructure is implemented will only happen when wider government reforms allow for more transparency and budget accountability.

7.12 Conclusion

This chapter has provided a review of some of the main institutional reforms that have taken place in sectors that influence natural infrastructure implementation. Due to the integrated nature of ecosystem-approaches, institutional arrangements are important components of successful integration. To understand why these arrangements are established and the barriers and opportunities that they present, it is useful to look at the history of their development and to analyse them in the context of wider government reform processes. The water sector is now faced with increasing focus on market-based approaches that emphasise performance management and decentralisation to improve efficiency. To implement change within these arrangements, three recommendations are outlined, firstly, to understand the socio-political context of the water sector, recognising the unique social and physical condition of each water basin, secondly, to recognise that establishing perfect governance arrangements takes time and that strategic and problem-solving approaches are required to make changes in areas where appropriate responses can achieve results in a non-ideal world, finally, to recognise the importance of monitoring performance and to understand how best to design monitoring instruments that clearly communicate the benefits associated with natural infrastructure options. Unless these can be communicated well within existing governance frameworks, more widespread implementation will be hampered by the increasing drive for quantifiable results within public service delivery.

CHAPTER 8 Mechanisms for mainstreaming natural infrastructure into infrastructure planning

8.1 Overview

This report examines the potential mechanisms required to mainstream natural infrastructure into traditional infrastructure planning, with a specific focus on water infrastructure. The report is based on the premise that to stimulate interest in natural infrastructure requires a good understanding of the enabling mechanisms such as financing options, legal and institutional arrangements and processes such as catchment and infrastructure planning. This is important for building a financial and business case to encourage investment in and protection of natural infrastructure such as wetlands, forests and rivers (Talberth *et al* 2012).

Traditional water infrastructure development is generally on the rise globally for various reasons such as the need for more water storage, hydropower and water transfers (Krchnak *et al* 2011). Most of the water infrastructure development is however envisaged to take place in the developing world where inadequate water infrastructure is a major problem (Camdessus 2003). This presents both an opportunity and a challenge, in terms of the negative consequences of infrastructure development on the environment and the opportunities of undertaking a holistic approach to infrastructure planning and development.

The need to develop a more holistic approach to understanding water infrastructure is compounded by the fact that in many parts of the world, water infrastructure is crumpling, with most infrastructure not built to accommodate the current situation. As a result there is renewed emphasis on developing sustainable water infrastructure due to the increasing population and changes in consumption patterns. The principle of stationarity, which assumes that fluctuations in natural systems only change within a fixed range of variability, has played a key role in the design of current water infrastructure systems (Milly *et al* 2008). Stationarity does not however take into account changes brought into the system as a result of human disturbances. Most water infrastructure are not therefore flexible in their design and might not be able to cope with uncertainties brought about by climate change and variability. To build resilient water infrastructure, consideration should therefore be given to the other dynamics that are at play such as the role of natural infrastructure in providing ecosystem goods and services.

Sustainable water infrastructure has become a fundamental principle of building resilience into water infrastructure planning and development. Sustainable water infrastructure in this context refers to both the built infrastructure and the supporting natural systems such as wetlands, floodplains, groundwater aquifers and the watershed **(**Bolger *et al* 2009). The challenge of the approach to sustainable water infrastructure is that infrastructure is still largely viewed in terms of its engineering element, and the challenges of quantifying the value of natural infrastructure.

To fully embrace sustainable water infrastructure there are some key principles that need to be adhered to. Sustainable water infrastructure specifically requires water systems to be highly adaptable by incorporating an element of flexibility to address uncertainties, and a watershed approach that takes into account the

Current enabling mechanisms for mainstreaming natural infrastructure can be categories into four main types: -

- Planning mechanisms (e.g. river basin planning, strategic environmental assessment
- Financial mechanisms (e.g. infrastructure charges, PES, grants).
- Legal mechanisms (e.g. land use authorizations, licensing, conservation of resources).
- Voluntary arrangements between users to secure public goods and services (e.g. conservation easements or stewardship).

8.2 Planning mechanisms for mainstreaming natural infrastructure

8.2.1 River Basin planning

River basin planning is widely used as a strategy for sustainable management of water resources, in recognition of the critical role played by the environment in providing important ecosystem goods and services. Human activities such as unsustainable land use practices, over abstraction of water resources and general degradation of the watershed is however negatively impacting the sustainability of the ecosystem services that humans enjoy. This has led to intense competition for the scarce water resources as a result of increased demand attributable to changes in consumption patterns, population and climate variability.

8.2.2 Considerations for effective river basin planning

Some considerations in effective river basin planning include the following (Pegram *et al* 2010): -

- Effective stakeholder engagement process in developing the objectives and strategic options for river basin.
- Transparency in information sharing and decision making, with information in the public domain and made available by the CMA.
- Integrated management, which recognizes the inter-related nature of hydrological, ecological, social and economic systems, in line with the national water policy and legislation.
- Adaptive management, which requires flexibility in approaches to respond to unforeseen circumstances or inadequate management decisions.
- Institutionalization of the process by linking to existing structures, developing water sector participatory bodies and empowering stakeholders.

River basin planning is therefore a strategic response to identifying ways in which scarce water resources in a river basin can be sustainably managed to meet the needs of the competing users. This is very pertinent, because as the watershed is degraded the distribution, reliability, timing and the quality of water resources is compromised. As a result long term planning becomes critical to balance these competing needs and for understanding the trade-offs between developmental and environmental needs. In essence river basin planning is about recognizing the role of natural infrastructure in meeting human needs, and how to ensure that these infrastructure are sustainably managed.

Approaches to river basin planning have evolved quite significantly, but the focus has generally been on achieving multiple objectives such as food security, energy, transport and water supply. Basin planning is relatively broad high-level undertaking that results in a vision being identified for the basin and acts as the basis for more detailed planning. Understanding the river basin planning process may lead to effective integration of natural capital in infrastructure planning by:

- Assessing and prioritizing issues of concern to be managed within a basin.
- Deciding on the way in which these priorities should be managed to build resilience into the natural infrastructure overtime.
- Specifying the way in which different competing purposes (such as abstraction, hydropower, environmental flows) may be implemented.

Considering that basin planning is a high level process, the opportunity that it presents for natural infrastructure is at the level of setting a vision and action planning, but the implementation of the vision is often a major challenge. To overcome this, additional planning may be required to identify practical steps for implementing the vision. For example this may require the development of water infrastructure to meet current water demands or to improve its reliability. It’s at this practical level where incorporation of natural infrastructure becomes crucial.

8.2.3 Some key challenges of river basin planning

- Implementation of a river basin plan is often a major challenge, because of their broad nature that does not lend them for use in implementation specific programs on the ground, often requiring additional planning.
- Balancing the competing interests of a broad range of stakeholders can be very complex process. This requires river basin planning to be conducted by highly skilled personnel with excellent negotiations skills, and good understanding of stakeholder engagement processes. However in many river basins lack of capacity presents a major hurdle.
- Overlaps in boundaries and mandates can potentially derail the implementation of a well-designed basin plan. Political boundaries and watersheds do not overlap in most cases, which complicates the process of basin planning and affect its implementation. To overcome this challenge may require the establishment of a framework that may include a legal agreement or compact for different political jurisdictions to work together.

8.2.4 Strategic environmental assessment as a tool for mainstreaming natural infrastructure

Strategic environmental assessment is a decision making tool ensure environmental considerations are mainstreamed at the highest level of policies, legislation, strategies, plans, and programs (PLSPP) (Hirji & Davies 2009). This arose as a strategy for addressing the ineffectiveness of environmental impact assessment (EIAs). Because EIAs are designed to evaluate the impact of a project within a very specific context, it fails to incorporate the broader spatial and temporal related impacts that a project might impact on the landscape. SEA therefore seeks to ensure that a holistic approach is undertaken in evaluating the impact of a project on the environment and for decision- making.

SEAs have been used for different purposes depending on the context in which it is being applied. For example at the national and transboundary level, they have been used to develop a shared understanding of environmental issues, identifying issues for more detailed EIA assessment and developing investment plans (Hirji & Davies 2009). Other examples for which SEA could be used in relation to the water sector include; developing a national or water sector policy, enacting water legislation, drawing up river basin plans, establishing a basin institution, formulating and implementing a national water supply, irrigation or energy master plan, identifying hydropower. In essence, SEA is a tool for decision makers to understand and manage the trade-offs between development and environmental and social concerns within a given context.

It's therefore self-intuitive that SEA provides one of the most robust tools for mainstreaming natural infrastructure at a strategic level, where the best opportunity lay for developing guidelines, strategies and financing mechanisms. Once a decision has been reached such as the commissioning of a specific water infrastructure it becomes very difficult to effectively incorporate natural infrastructure considerations at the project level.

8.2.5 Traditional infrastructure planning

Even though the best opportunity to securing natural infrastructure is at the strategic level of policies and strategies, traditional infrastructure can be modified to be more effective in protecting natural infrastructure. According to a recent study by Mathews *et al* 2011, natural infrastructure considerations can be incorporated at 3 key levels when planning for water infrastructure (Figure 8.1):

Project identification phase. At this very early stage, consideration should be given to whether such water infrastructure is needed in the first place, especially if it is going to be a large water infrastructure that will have massive environmental and social implications. Issues to consider at this stage is optimization in the use of current infrastructure through effective water conservation and demand management, or building the infrastructure in stages as climate change trends become more clear to build resilience into the infrastructure (Mathews *et al* 2011).

Implementation phase. An important undertaking at this stage should be the explicit incorporation of natural infrastructure such as ecosystems into the infrastructure development (Mathews *et al* 2011). This can be achieved using several interventions such as floodplain restoration, or the use of bioshields for flood management.

Operations and maintenance phase. Existing infrastructure also lend themselves to modification, by incorporating flexibility into their operations to take account of uncertainty brought about by climate change. Existing infrastructure could also be modified, through a process of redesign by retrofitting, relicensing and reassessment of the projected and incurred environmental impacts (Mathews *et al* 2011).

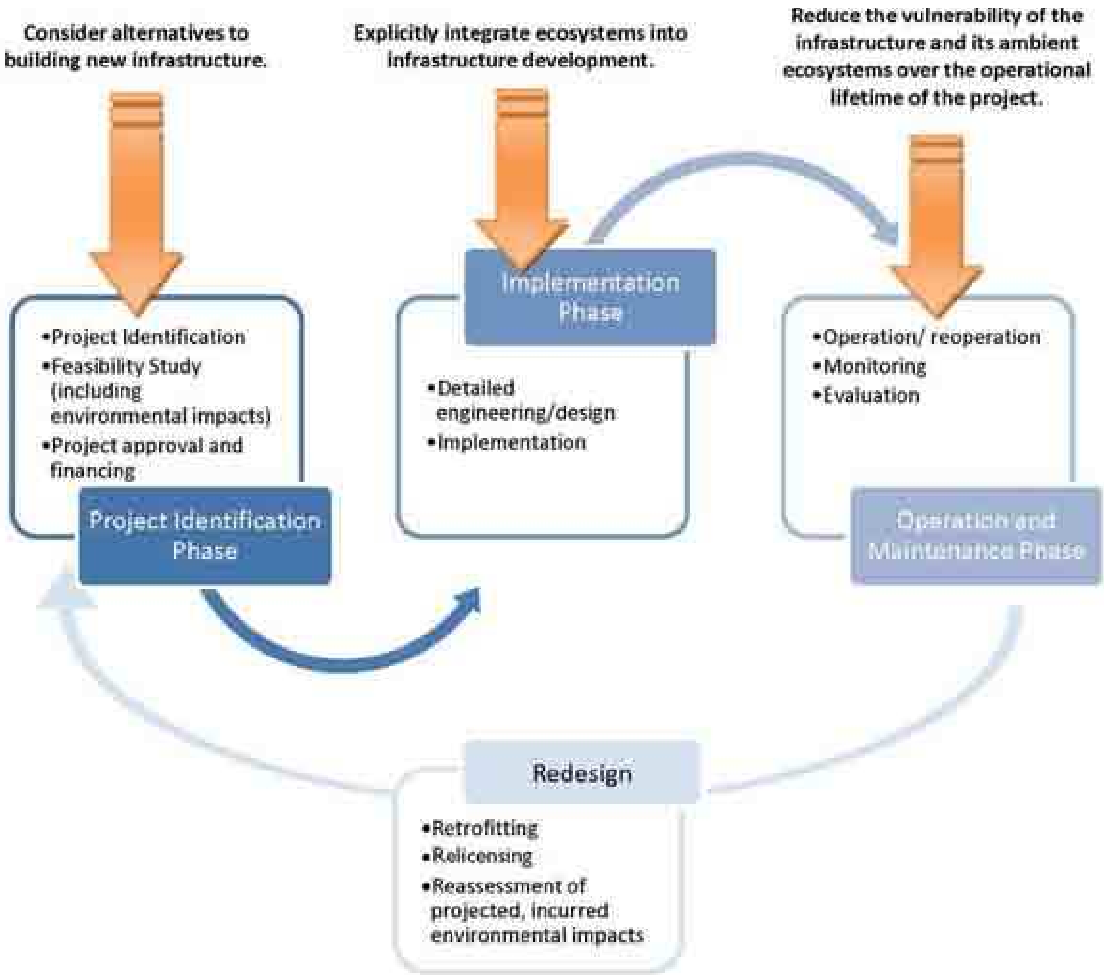


Figure 8.1. Proposed actions for how natural infrastructure can be mainstreamed into engineering water infrastructure (source: Mathews *et al* 2011).

8.2.6 Key issues associated with planning engineering infrastructure

- The long term strategic planning required for developing water infrastructure offers a good opportunity for holistic approach that effectively incorporates natural infrastructure.
- Effective real options analysis presents useful opportunity for analyzing the cost effectiveness of incorporating natural infrastructure, and how to modify existing infrastructure.
- Most of the infrastructure planning are too engineering focus and barely make consideration for environmental and social dynamics to be understood
- There are no well-established metrics and frameworks for explicitly incorporating natural infrastructure into traditional infrastructure planning.

8.3 Financial mechanisms

Public sector infrastructure finance is almost always concerned with three types of investment: new build to cater for increased demand and extension of services, refurbishment and backlogs to cater for existing users and overdue investment; and operations and maintenance (O&M). Each of these may attract different funding sources and require different financing mechanisms.

Within the water sector there are two distinct categories of infrastructure – water resource infrastructure, and water services and sanitation infrastructure, including wastewater management. Once again, the financing mechanisms may differ for the two categories, due to the different institutional environments and different funding sources. However they are also inter-linked through the value chain for water.

Regardless of the type or category of infrastructure, there are only three ways to pay for it – taxes, transfers (grants, donations) and tariffs. The 'three T's' were put forward by the World Panel on Financing Water Infrastructure (the "Camdessus Panel", established in 2001). A critical determinant of which one takes precedence is the extent to which the infrastructure is of a social or economic nature. Economic infrastructure is infrastructure where the investment can be recovered from users (tariffs), whereas social infrastructure will be reliant on fiscal funding (taxes).

Whilst the sources may be limited, there are a variety of mechanisms that can be employed to match the cashflow of these sources to the cashflow required to fund the establishment (and subsequent operations and maintenance) of the infrastructure. These mechanisms include the use of debt and equity from a range of institutions (private, public, multi-lateral, etc.). Private sector involvement can range from equity investment to long-term concessions.

For this review, financial mechanisms for enabling sustainable water infrastructure were assessed as follows: 1) current approaches to financing water infrastructure, specifically hard engineering infrastructure at the global, regional and national level 2) traditional streams of funding such as private finance, infrastructure charges, and 3) innovative financing mechanisms such as payments for environmental services. The emphasis of this analysis will be on trying to identify the potential opportunities where natural infrastructure could benefit through such financing mechanisms.

8.3.1 Financing water infrastructure at the global scale

Multilateral and bilateral agencies (Overseas development aid) are traditionally the major sources of funding for large water infrastructure such as dams, especially in developing countries. Due to the environmental and social impacts of large dams however, there is a general reluctance of donor agencies to fund these kinds of projects, with exception of water provision and sanitation.

Funding by global players is not immune to other external drivers, the recent global economic crisis for example had a major impact on infrastructure financing, The World Bank is estimating a finance gap of $270 billion per year in infrastructure funding. While other large existing infrastructure will experience a shortfall of close to $70 billion mostly in developing countries. It is therefore clear that relying on global finance alone for funding infrastructure can be risky.

In terms of water infrastructure the World Bank estimates that 900 million people globally don't have access to safe drink water. The challenge therefore remains enormous in trying to meet the Millennium Development Goals (MDGs). In an attempt to meet the infrastructure challenges faced globally, between 2003 and 2008 the bank's lending to infrastructure related spending grew by almost 88 percent. This expenditure was further scaled up when the World Bank through its **Infrastructure Recovery and Assets (INFRA) platform** and International Finance Corporation's (IFC) **Infrastructure Crisis Facility** (ICF), launched a $55 billion facility to finance infrastructure projects developing countries for a period of 3 years.

The EU on the other hand, supports 60 developing countries in the water sector, most of them located in Africa and Asia. The focus of the support is on water supply and integrated water resource management in addition to political engagement. In 2006 the EU established the EU-Africa Partnership for Infrastructures. This project among others focuses on the management of cross-border catchment areas, flood defense, capacity building and water resource monitoring.

8.3.2 Financing infrastructure at the regional level

Key players in financing water infrastructure at the regional level are through development banks and bilateral arrangements. These entities play a critical role in financing out of budget projects, and borrowers that cannot access funding from commercial banks (OECD 2010). Financing from development banks is structured similarly to commercial banks, but generally have lower interest relative to the market, have longer time frames and customized repayment periods (OECD 2010). In addition providing financing, most development banks offer advisory services, which are structured as grants and loans to their clients to support project planning, implementation and its sustainability. Some of the major players include: -

- The Inter-American Bank,
- The African Development Bank (AfDB),
- The Asia Development Bank (ADB).

Apart from development banks, bilateral agencies that are key for financing infrastructure at the regional level may include entities such as Agence de Francaise (AFD), United States International Development Agency (USAID) and *Gesellschaft für Internationale Zusammenarbeit* (*GIZ*). For example between 2009 and 2010 AFD spent close to €500 million for financing water infrastructure (Fig. 8.2).

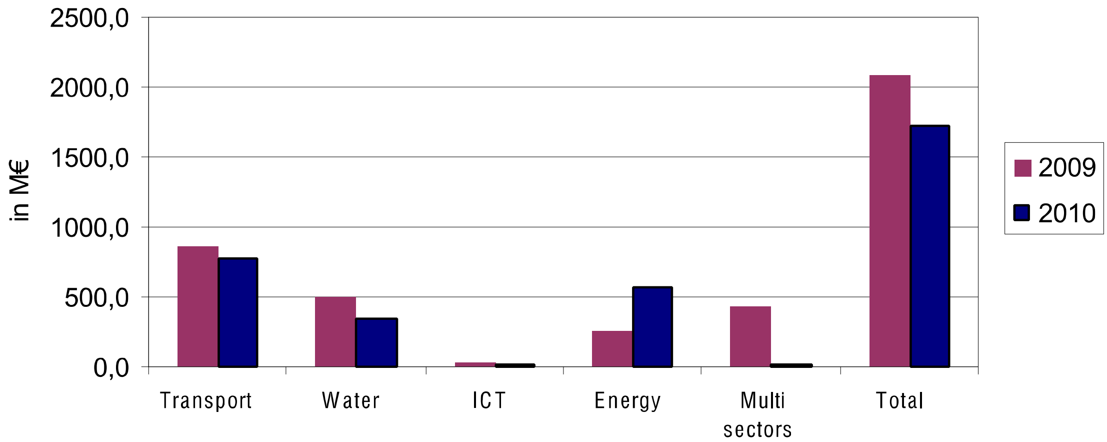


Fig. 8.2. AFD Commitment in Africa by nature of infrastructure (Moineville 2011).

8.3.3 The role of national governments in financing water infrastructure

National governments remain the main sources of funding water infrastructure globally. This means that the burden of water financing still lays mostly with the tax payers not users, resulting in perverse outcomes such as poor sense of water conservation and demand management, requiring more investment in water infrastructure. Developing countries have therefore generally relied on public expenditure to finance water infrastructure, specifically capital cost. Developed countries on the other hand, have well devolved funding infrastructure models that rely extensively on well functioning local authorities who are responsible for raising the finance for water infrastructure through market based mechanisms such as borrowing from commercial banks and issuing of Municipal bonds.

In South Africa for example most infrastructure is financed through public funding (Fig. 8.3). The public sector funding is generated through infrastructure grants and water sector pricing (charges, tariffs, levies). For first generation water infrastructure the funding was primarily from the national budget, moving forward however other innovative funding mechanisms might need to be explored (Ruiters 2010).

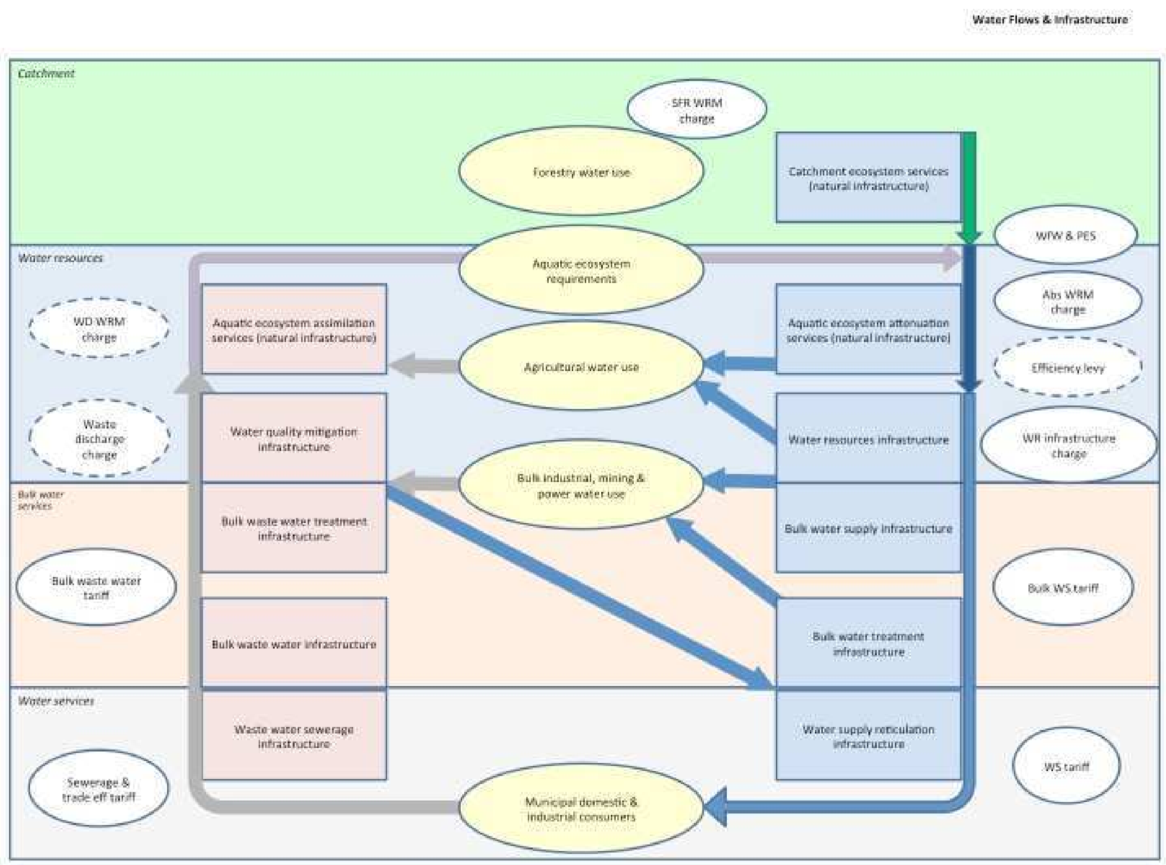


Fig. 8.3: Funding flows for water resource management in South Africa

At the local level Water Boards can levy tariffs on water users, which in some countries like South Africa are legal entities and financially self-sufficient. Such water tariffs are levied on local Municipalities for the supply of bulk raw water. In relation to funding water infrastructure, water tariffs at the local level primarily serve the purpose of financing regional water infrastructure.

8.3.4 Private sector financing

The private sector is a key player in the financing of water infrastructure, but unfortunately the water sector is still viewed as a high-risk area and hence unattractive to funders. The risks in the water sector are associated with the perception that return on investment is not immediate, which requires long-term commitment. Even though the return on investment in the water sector can be guaranteed for a long time, private sector participation in the water sector is generally disappointing. In a World Bank Study on the track record of private-public partnership, it was found that private sector investment in water infrastructure is generally limited compared to other infrastructure, representing 5.4% of total investments between 1990- 2000 (OECD 2010). The role of the private sector in the water sector is somewhat different for Organization for Economic Cooperation and Development (OECD) countries, which has seen an increase in participation by the private sector in water services. This represents a trend in the private sector whereby private investors are opting for low risk short-term arrangements with minimal investment. In absence of the international private finance, some strong regional players have emerged to fill this void in servicing the water sector, leading to successful establishment of public utilities across the developing world in countries such as Cambodia, Uganda and Mozambique.

8.3.5 Innovative financing mechanisms: Payment for Environmental Services

Payments for environmental services are innovative funding mechanisms designed to recognize the goods and services that accrue from the environment. Watersheds provide important services, which include provisioning services (e.g. water supply) regulatory services (e.g. flood attenuation) supporting services (e.g. biodiversity habitats) and cultural services (e.g. aesthetic enjoyment) (Table 8.1). Due to the very nature of watersheds however, these services are seldom valued because they lie outside the domain of markets (Postel & Thompson 2005). This view is starting to change with the advent of the concept of payments for ecosystem services.

Table 8.1: Ecosystem goods and services provided by healthy watersheds (Postel and Thompson 2005).

- Water supplies for agricultural, industrial, and urban-domestic uses
- Water filtration/purification
- Flow regulation
- Flood control
- Erosion and sedimentation control
- Fisheries
- Timber and other forest products
- Recreation/tourism
- Habitat for biodiversity preservation
- Aesthetic enjoyment
- Climate stabilization
- Cultural, religious, inspirational values

There are several mechanisms for implementing payments for environmental services (PES) at the watershed level as a form of economic incentive to secure the critical services provided by the watershed. Payments for watershed services usually involve downstream beneficiaries making a payment to upstream landowners as an incentive to protect the watershed**.** These types of payments can be categorized into three main types (Hanson 2011 *et al* 2009; Greiber 2009):

- Private transactions
- Cap and trade transactions
- Payments made to generate public benefits

8.3.6 Private transactions

The private transactions referred to here are voluntary payments made by downstream beneficiaries of an ecosystem service to upstream providers of the service. This typically involves paying landowners upstream to maintain the watershed in such a way as to avoid any negative impact on downstream water users such as altered water quality, reduction in stream flow or flooding. Another characteristic of these private PES transactions is that in some cases there is cost sharing among the private parties involved and if a land purchase is involved upstream, it may be leased back to the owner with the objective of ensuring the protection of the watershed (Greiber 2009). In cases where the transaction with the upstream landowner does not involve leasing the land, they may get paid to undertake restoration activities on the landscape such as riparian protection and changes in agriculture practices.

8.3.7 Cap and trade transactions

These types of transactions are based on existing rights such as pollution or abstraction limits, and a trading scheme is then established to trade those rights. In these transactions credits maybe issued by an authority to a particular individual who engages in an activity that results in a watershed protection, such as pollution control. The individual who owns the credits can then choose to sell them to any person who is embarking on an activity that might result in them exceeding the limits of the pollution requirement as set by the regulator. There are several examples of cape and trade schemes globally, mostly notably the United States Clean Water Act, which has a wetland-banking scheme. The wetland banking scheme requires landowners who damage wetlands to offset that by restoring and protecting other wetlands either on site or elsewhere. In this approach players therefore have to purchase credits in order to meet their mitigation obligations (Greiber 2009). Cap and Trade schemes can be rather complex, requiring some clear guidelines as outlined by Greiber 2009 below: -

- Clear definition of those activities that have a negative impact on ecosystem services and thus lead to mitigation obligations;
- Development of transparent standards to quantify the unit of exchange (e.g. based on their actual value and/or function, or based on the size and/or geography of the concerned land);
- Determination of units of restored, created, enhanced or preserved ecosystem services which will be converted into tradable credits;
- Establishment of procedural frameworks for opening, managing and closing mitigation banks, for protecting the resulting ecosystem services in perpetuity, and for ensuring fair trade;
- Creation of insurance and liability systems to guarantee long-term offsetting and stewardship success.

8.3.8 Payments made to generate public benefits

Payments that are made for public benefits constitute an arrangement where a government entity is involved, and may include collecting fees, land purchase or granting of rights to use land resources (Greiber 2009). These arrangements mostly involve Municipalities, local government and utilities. This is the most common form of PES, because of the simplicity of its set up, where the public entity is the sole buyer or seller of the ecosystem service.

In cases where the government is the purchaser of such ecosystem service, it may take the form of engagement that ensures the protection of 'public goods and services'. For example government may pay landowners to protect a watershed to yield benefits to the general public as opposed to designated groups such as those involved in private PES transactions. Public goods are generally under funded because they are benefits that are enjoyed by all, and watershed payments can be a useful mechanism for boosting such areas (Hanson *et al* 2011).

There are several examples of regulator driven PES schemes, such as the widely implemented land stewardship programme under the United States Department of Agriculture (USDA). Under this scheme private landowners sign agreements the government authorities to ensure that they conserve the natural resources in their private lands, such as wetlands, soils, floodplains and forests. The government will in-turn pay the landowners for undertaking such conservation activities.

Another widely cited example of is that of the Catskills watershed in New York City, where the authorities opted to conserve the upstream watershed as an alternative to building additional water treatment plants. To-date more than $1.5 billion has been spent by the city to sustain the critical water supply services provided by the Catskills watershed. The payments in this scheme are directed towards forest conservation, habitat rehabilitation and the creation of green corridors to link up reservoirs. Investing in this green infrastructure turned out to be way cheaper than the construction of a water filtration plant.

8.3.9 Voluntary *n*on-financial mechanisms for conservation

Most voluntary mechanisms for conserving natural capital have some element of financial compensation (Rissman and Syre 2011), as discussed under the financial mechanisms. However, there are cases where private landowners voluntarily give up their land for conservation purposes, without expecting any compensation. This may be in the form of donating the land itself to a conservation organization, where for example in South Africa, the World Wide Fund (WWF) for Nature, owns large tracks of land, which has been voluntarily donated by private landowners through endowments for conservation purposes. In some cases landowner may enter into a long term agreement that prescribes specific land use activities but the owner retains the land tittle, and therefore ownership of the land.

Voluntary mechanisms for conservation can also take the form of government-assisted community programs. In this case the government works with a local communities to protect critical ecosystems, raise funds to purchase land or make arrangements for co-management of conservation lands. Government assistance to such community conservation programs may include provision of money, training or the setting up of conservation groups (Stoneham *et al* 2000).

8.4 Legal mechanisms to ensure protection / consideration of natural infrastructure

The legal mechanisms for protecting natural infrastructure vary widely in different parts of the world, which can however be broadly categorized into global regional and national legal frameworks.

8.4.1 Compliance with international and regional environmental agreements

Many countries have international obligations for which they have committed and would need to meet their international obligations. Some key international conventions such as the international convention on wetlands (Ramsar) and the Convention on Biological Diversity (CBD), have obligations that need to be met by signatories such as the listing of wetlands of international importance under Ramsar and the requirement by CBD for countries to develop national biodiversity strategy and action plans (NBSAPs), all provide opportunities for protecting natural infrastructure. In an assessment carried out by in the Pacific Islands (Rissman & Sayre 2011), they however found no evidence that countries in that region are meeting their international obligations under the various conventions that they have committed. The poor record of meeting international obligations is attributable to lack of capacity in that region and the same can be said of most developing countries where both financial and human resource can be a major constraint. In the developed world, the lack of compliance in meeting international agreements could be attributed to the political implications for such a decision. For example, the recent introduction of carbon taxes in Australia is going to be a major topic of political debate in that country, with major implications for elections.

8.4.2 National level legal instruments

There is a broad range of legal instruments at the national level that govern natural resources management. Even though these arrangements vary broadly, mechanisms for natural infrastructure range from the constitution to legislations, policies and strategies and action plans.

8.4.3 Constitutions

National constitutions are the highest level at which natural resources can be protected, as evidenced by some countries that have declared the right to a healthy environment a human right (RSA 1996). The national constitution is however a very broad legal instrument that defines a nation by asserting its sovereignty, powers and basic legal principles (Greiber 2009). Due to the very nature of the constitution, they seldom make references to specific natural infrastructure, such as ecosystems that need to be protected. But in some cases like the South African constitution, the bill of rights clearly stipulates the following “everyone has the right to an environment that is not harmful to their health or well being, and to have the environment protected, for the benefit of present and future generations, through reasonable legislative and other measures” (RSA 1996). In this case the constitution makes a clear provision for sustainable development by ensuring that natural resources are managed sustainably for future posterity. Even though many countries may not make such explicit reference to environmental protection in their constitutions, they have some form of legislation that governs environmental protection.

8.4.4 National legislations

National legislations are the vehicle through which a country can put into effect the provisions and rules that have been broadly outlined in its constitution, and it is through these legislations where provisions for supporting natural infrastructure are clearly defined. The legal mechanisms that may arise could include the following: -

- **Conservation of resources**. These legislations that govern natural capital such as biodiversity or water resources. Under this category various acts that govern natural resources protection can be defined, such as the establishment of protected areas and the implementation of environmental flows.
- **Land authorizations**. These legal mechanisms that define land use activities in particular area to ensure a strategic approach to how land resources are managed, for example areas that are designed for agriculture may not be converted into a residential area unless a specific procedure is followed authorize a change in the zoning of the landscape.
- **Licensing mechanisms**. These are provisions that require sustainable management of natural resources through the implementation of user licenses such as water use license for big users, or the issuing of mining license. Licensing may make provision for specific conditions that minimize environmental impact to be met, such as the undertaking of an environmental impact assessment (EIA).
- **Integrated planning**. These are legislative measures that recognize the interconnected nature of the landscape, and the need to balance economic, environmental and social needs at a strategic level. Under these legislation, it might require integrated planning at the local level through the development of integrated development plans (IDPs), for Municipal authorizes to ensure that a holistic approach is taken in trying to meet development goals at the local level.

It is important to note that national legislations give rise to mechanisms that sit in different mandates, which presents both challenges and opportunities at the same time. In many countries for example the water and environmental mandates sit under different mandates, without effective cooperative governance and policy coherence, such arrangements may prove to be a major constraint to effective management of natural infrastructure.

8.5 The role of resource owners

Landowners play a critically important role in supporting natural infrastructure, because a lot of land of conservation value is private hands, and the attitude and willingness of landowners in supporting natural infrastructure become paramount. For example the voluntary mechanisms outlined above and payments for ecosystem services cannot be implemented without willing landowners.

Under this mechanism, the key drivers are related to issues of property rights and how that might influence private landowners to participate in voluntary mechanisms for protecting natural infrastructure. Some countries have legislation that prescribes the responsibility of landowners towards securing their pieces of land. In South Africa for example the Conservation of Agricultural Resources Act (CARA) requires private landowner to remove alien invasive species from the properties, particularly those that contribute to streamflow reduction along riparian areas.

8.6 Conclusions

Natural infrastructure supply important ecosystem services and goods that are critical for the survival of humanity, and this important role is increasingly getting recognized. But challenges of incorporating natural infrastructure fully into developmental frameworks still remain. Developing understanding of the possible mechanisms for mainstreaming natural infrastructure, especially those related to water security is vital because of the potential global threat on water security, as populations increase compounded by climate change. This report sought to broadly sketch out some of the key mechanisms for supporting natural infrastructure and their role in providing water security, ranging from planning to the implementation phase, and identified some levers where natural infrastructure can be supported. The key mechanisms identified are broadly categorized into planning, financial and voluntary and legal mechanisms. Each of these mechanisms present a varying degree of complexity in the ease with which natural infrastructure can be mainstreamed, but the key message is that to effectively mainstream natural infrastructure requires a multiple pronged approach to identity opportunities by taking advantage of all the mechanisms available at ones disposal. It was also apparent from the review that progress has been made in understanding the role of natural infrastructure, and there are innovative case studies being implemented globally, the key challenge however is scaling up these innovative approaches in-order to bring about large scale change on the ground. Developing understanding on the type frameworks required to scale up innovative approaches both horizontally and vertical could therefore be a useful way forward in developing support for natural infrastructure.

CHAPTER 9 The Extent of Incorporation of Natural Infrastructure into IWRM

9.1 Introduction

Water managers have to deal with increasingly complex water issues. It is difficult to think of a more crucial resource to human life than water. It is known that a strikingly small fraction (0.26 percent) of the world's freshwater resources is readily accessible for human use (Shiklomanov, 2000). Nevertheless, water is one of the most overexploited and deteriorated natural resources. The majority of rivers in the world are severely affected by anthropogenic activities. This is manifested through the alarming decline of freshwater species. For example, it is documented that 27 percent of North American freshwater species are endangered (Gleick, 2003). Substantial flows in many waterways such as the Nile, Huang He (Yellow), Amu Darya and SyrDaria and Colorado rivers no longer reach their deltas (Gleick, 2003). Both developing and developed countries suffer hydrological extremes—such as droughts and floods—which are projected to become more frequent and severe due to climate change. Furthermore, meeting the basic human need for water remains a massive challenge in many countries around the globe. More than one billion people lack access to drinking water and around 2.4 billion suffer from poor sanitation (Gleick, 2003). Notably, in India more than 600 million people do not have access to toilets, and around 97 million of them live in a water-stressed region (Narain, 2012). These ecological and social impacts are striking evidence that the current approaches to water management fall short in achieving a sustainable water use. The prevailing approaches to water resource management in the past century—engineering approaches (wastewater treatment plants, dams, dykes, *etc*.)— were developed to provide various significant benefits to people. For example, reservoirs, created by dams, serve multiple beneficial purposes, including irrigation for food production, water for consumption, hydroelectricity, flood control, and navigation. However, technical approaches to water management lacking careful consideration of ecological processes have negatively impacted ecosystems, undermining nature's capacity to generate not only water-related goods and services such as water provision, water purification, flood control, and climate stabilization but also carbon sequestration, fisheries, erosion and sediment control, and recreation (Postel &Thompson, 2005) . There is ample evidence of river impoundment causing significant environmental harm ranging from the disruption of natural river flows to greenhouse gas emissions, from habitat loss to species extinctions (WCD, 2000; Nilson *et al.*, 2007). Furthermore, impoundment has numerous negative social impacts, especially to downstream communities dependent on fisheries and flood-based agriculture for their subsistence (Richter *et al.*, 2010).

Hard infrastructural measures generally exhibit the high maintenance costs and the risk of degraded performance over time (WWAP, 2012). The deficits in Canada's municipal infrastructure are a case in point. Shockingly, more than 60 percent of the nation's wastewater treatment facilities were unfit to continue operating in 2003 (Sandford&Phare, 2011). Financially speaking, billions of dollars of investment in water and wastewater facilities are required (30- 40 billion for the province of Ontario alone) to remedy this situation. According to the Canadian Water Network, Canada's infrastructure maintenance deficit is $88.4 billion (Sandford&Phare, 2011). Clearly, Canadian municipalities can save money if they improve not only their built infrastructure, but also natural infrastructure to resolve their water issues, in other words, if they incorporate 'green' solutions into their water management practices.

Given the maintenance costs of built infrastructure and looming challenges in water distribution, there is heightened interest in using natural infrastructure solutions for water management (WWAP, 2012; Coates & Smith, 2012). In essence, natural infrastructure solutions are measures directed to enhance ecosystem functioning and the provision of ecosystem services (Coates & Smith, 2012). They are based on the understanding of the imperative role of biodiversity and ecosystems in the water cycle (Coates & Smith, 2012). Ecosystem protection enables nature to regenerate; therefore, ecosystem preservation presents a natural infrastructure solution on its own. There is growing scientific evidence that shows the pivotal role of biodiversity in the hydrological cycle (Coates & Smith, 2012). Ecosystem functioning, and inevitably, the provision of ecosystem goods and services depend on biodiversity. Forests (and other land-cover types), soil, and wetlands are major components of ecosystems, and can be considered as natural water infrastructure (Coates & Smith, 2012). There is increasing scientific evidence supporting the critical role of biodiversity in ensuring water security:

- The role of wetlands in water purification/filtration/ flow regulation, etc.
- The importance of good soil quality in erosion and sediment control, groundwater recharge
- The role of land cover in water supply (through local climate regulation), soil stability, etc. (Coates & Smith, 2012; Jewitt, 2002; Ashton *et al.*, 2005).

Many researchers highlight watershed protection as a long-term sustainable solution to complex water problems, thanks to the array of valuable ecosystem goods and services that healthy watersheds provide (Postel& Thompson, 2005;Jewitt, 2002; Turner &Daily, 2008). Natural infrastructure solutions are cost effective options for water security. Residents of Colombia's capital, Bogota, receive the majority of their drinking water from a wetland located in the Chingaza National Park. The water is protected by the public water utility "with minimal treatment" (Postel& Thompson, 2005). This remarkable natural infrastructure "absorbs, filters and releases" water of drinking quality of sufficient quantity per unit of time "with little seasonal variation" (Postel& Thompson, 2005). This helps significantly reduce the water treatment costs. Postel and Thompson (2005) illustrated this finding in industrial countries' watershed protection. They showed the results of the analysis of 27 U.S. water suppliers and pointed out that the costs of obtaining drinking water from watersheds covered by at least 60% forest were half that of 30% watersheds and one-third that of 10% ones (Table 9.1).

Table 9.1. Forest cover and predicted water treatment costs based on 27 U.S. water supply systems*.Source: Adapted from Ernst (2004). In Postel& Thompson, 2005.
*Based on treatment of 22 million gallons (83,270 m3) per day, the average daily production of the water suppliers surveyed.

Share of watershed forested	Treatment costs per 3,785 m^3	Average annual treatment costs	Cost increase over 60% forest cover
60%	$37	$297,110	–
50%	$46	$369,380	24%
40%	$58	$465,740	57%
30%	$73	$586,190	97%
20%	$93	$746,790	151%
10%	$115	$923,450	211%

It seems to be an increasing trend that many U.S. cities are shunning expensive filtration plants in favor of lower expenses on watershed protection (Table 9.2).

Table 9. 2. Selected U.S. cities that have avoided construction of filtration plants through watershed protection . Adopted from Postel&Thompson, 2005.

Metropolitan area	Population	Avoided costs through watershed protection
	(thousands)	
New York City[a]	9,000	$1.5 billion spent on watershed protection over 10 years to avoid at least $6 billion in capital costs and $300 million in annual operating costs.
Boston, Massachusetts[b]	2,300	$180 million (gross) avoided cost.
Seattle, Washington[c]	1,300	$150–200 million (gross) avoided cost.
Portland, Oregon[d]	825	$920,000 spent annually to protect watershed is avoiding a $200 million capital cost.
Portland, Maine[d]	160	$729,000 spent annually to protect watershed has avoided $25 million in capital costs and $725,000 in operating costs.
Syracuse, New York[e]	150	$10 million watershed plan is avoiding $45–60 million in capital costs.
Auburn, Maine[f]	23	$570,000 spent to acquire watershed land is avoiding $30 million capital cost and $750,000 in annual operating costs.

Notes:
[a] The City is currently being required to construct a $687 million filtration plant for the more-developed Croton watershed, which supplies about 10% of the city's water. The filtration waiver applies to the Catskills/Delaware watershed, which supplies about 90% of the city's water (NRC, 2000).
[b] *US v. Massachusetts Water Resources Authority* (2000).
[c] Supply from Seattle's Cedar River watershed is unfiltered, but that from the Tolt watershed is now filtered (Flagor, 2003).
[d] Reid (2001).
[e] ECONorthwest (2004).
[f] Ernst (2004).

More water managers are acknowledging the cost-effectiveness, achievement of long-term goals and co-benefits associated with the improvement of various ecosystem services, such as recreational benefits, greater biodiversity and higher crop and fish yields. In fact, there are many cases where the protection has been established because of safeguarding water supplies alone. For instance, the World Bank loaned Indonesia $1.2 towards the establishment of Dumoa-Bone National Park. This loan was granted due to the benefits the park had on irrigation projects in the lowlands (Turner&Daily, 2008). In Honduras, protection status was given to La Tigra National Park due to the benefits it provided; namely, its cloud forests led to the generation of 40 percent of their capital's water supply, for around 5 percent the cost of other alternatives (Postel & Thompson, 2005).

This chapter assesses to what extent these natural infrastructure solutions are implemented in integrated water resources management.

9.2 Integrated Water Resources Management

The concept of Integrated Water Resources Management was developed in response to the growing recognition of the importance of a holistic approach to water management—an approach where land and water resources can be managed together. As defined by the Global Water Partnership, IWRM is a *"process that promotes the coordinated development and management of water, land and related resources in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems."* IWRM views water management at the basin level. According to the IWRM principles water is recognized as a finite, vulnerable resource. Its nature prompts efforts of water conservation, water efficiency measures and managing the demand side of water resources. In IWRM water is also recognized as an economic good. Moreover, IWRM promotes a participatory approach involving users, planners and policy makers at all levels, and allocates a special role to women in water management and the allocation of water resources.

IWRM incorporates scientific and technical elements into water management, as well as "socioeconomic components (often referred to as water governance and including such elements as institutions, regulations, policy, public awareness, political realities)" (WWAP, 2009).
IWRM strives to achieve economic efficiency in water use, equal access to water resources and environmental and ecological sustainability. In order to accomplish these goals, the cross-sectoral integration of the following, *inter alia*, is crucial: 1) a general framework consisting of policies, legislation, and information; 2) the institutional roles and functions various administrative levels and stakeholders; and 3) management instruments for regulating, monitoring, and enforcement for decision-makers (Jønch-Clausen, 2004). Moreover, IWRM stresses the importance of water for maintaining ecosystem services.

9.3 IWRM in practice

Many countries of the world face serious challenges realizing IWRM in practice. The most recent UN- Water survey assessing the status of implementation of integrated approaches to water resources management indicates that countries are significantly behind the target set in the Johannesburg Plan of Implementation. It shows that 65 percent of 134 surveyed countries have developed integrated water resources management plans and only 34 percent indicate an advanced stage of implementation (UNEP, 2012). However, the survey also suggests that the IWRM approach is "being seriously pursued by governments in many countries" worldwide (UNEP, 2012).

Water experts highlight a growing concern that "although considerable attention has been given to human water needs for drinking water and sanitation, food production and industry, less attention has been devoted to the 'environmental and ecological sustainability' element, the ultimate focus of which is sustainable ecosystem services" (WWAP, 2009). Furthermore, the authors argue that the neglect of the essential components which provide ecosystem services could put the entire water sub-sector at risk. It is therefore easy to see the benefits of effectively managing ecosystems (improved human health, well-being and economic stability to name a few). However, as seen in many cases, economic development often overshadows environmental concerns, leaving them unincorporated and inadequately addressed (WWAP, 2009).

Furthermore, it is well-documented that today's water resource management initiatives fail to take all relevant ecosystem services into account. This inevitably leads to management efforts focusing on selected services. To remedy this issue, IWRM must balance all current ecosystem services as well as consider future ecosystem services, by improving factors such as ecosystem resilience (WWAP, 2009).

In fact, due to the undervaluation of ecosystem services by most water managers, many of them degrade over time. Additionally, mainstream water management tends to focus on individual concerns—including water pollution control, water supply and allocation and specific targeted water-use sectors—instead of viewing them collectively (WWAP, 2009).

Managing ecosystem services by ensuring that ecosystems have sufficient water of adequate quality is key to achieving water security and promoting human health and well-being. Natural infrastructure solutions are increasingly being applied to effectively tackle water problems worldwide.

9.3.1 The level of attention to ecosystems in IWRM

Analyzing the incorporation of natural infrastructure solutions in IWRM provides some interesting insights into the question of environmental sustainability in IWRM. The question of the extent of incorporation of natural infrastructure solutions inevitably addresses the deeper question of how ecosystem protection is viewed in practice and in theory. Ecological sustainability is one of the key objectives of IWRM. Indeed, IWMR proponents promote IWRM as an effective framework that is able to ensure sustainability of water resources. The UN-Water report on the status of implementation of integrated approaches to water resources management shows that some countries identified positive impacts of IWRM on the environment (UNEP, 2012). For the past 20 years, the greatest positive environmental impacts from improved water resource management have been recognized mainly by countries with a high Human Development Index (HDI). The results of the survey indicate that on average, these countries gave a rating of 4 (on a 1 (low) to 5 (high) scale) to these impacts. On the other hand, 24 percent of countries with a low HDI gave the lowest possible rating, with none giving the highest. However, countries across all measures of HDI were able to recognize the positive impacts of improved management, as well as decreasing rates of ecosystem degradation and improvements in environmental flow. Among all countries, the most commonly recognized positive impact is improved water quality. This often comes as a result of improved waste treatment, especially in less developed countries such as Rwanda. Finally, improvement in flood/drought management and prevention has been acknowledged by many countries such as Cuba, Portugal and Ghana, where changes which recognize climatehave been implemented on a national level.

While in theory IWRM holds much potential for ecological sustainability and the enhancement of ecosystem services, its critics state that in practice, there is a limited or absent emphasis on ecosystem protection. One striking example is the most recent annual report of Global Water Partnership, the organization responsible for the implementation of IWRM. In discussing its progress towards the achievement of its goals, it rarely even mentions ecosystem protection. To be precise, a simple keyword search yields only one mentioning of biodiversity and only nine repetitions of the word 'ecosystem'. Moreover, the UN-Water 2012 survey on the status of IWRM worldwide showed that 50 percent of the 134 responding countries ranked water for the environment as medium to low priority, with a small percentage ranking it as 'not a problem.'This is in line with Rahaman&Varis's (2005) criticism of IWRM for the lack of focus on river restoration: "IWRM principles do not clearly focus on or address the mechanism of river restoration, which is necessary for the sustainable water resources management in areas that have undergone or are presently subjected to notable modifications."

Some water experts point out that the focus of IWRM is predominantly on water allocation and pollution, not on ecosystem health (WWAP, 2009). Water expert MogensDyhr-Nielsen provides a vivid example of this focus using a case study of IWRM implementation in the Bang Pakong River Basin (Thailand) (WWAP, 2009). Additionally, a large number of IWRM peer-reviewed articles and case studies presented in GWP ToolBox concentrate on institutional reforms, the establishment of networks, and the achievement of multi-stakeholder participation. Little emphasis is placed on ecological aspects in IWRM.

There are also IWRM case studies that highlight ecosystem preservation; however, the main goal of ecosystem protection in these cases is wildlife conservation, not the amelioration of water-related ecosystem services. The Puerto Rico IWMR case study (below) vividly illustrates this point.

9.3.2 The level of attention to natural infrastructure solutions in IWRM

Overall, a literature review suggests that there are limited examples of the utilization of natural infrastructure solutions in IWRM worldwide. Not surprisingly, natural infrastructure solutions prevail in IWRM projects that are led by conservation organizations, such as the International Union for Conservation of Nature, Nature Conservancy, or World Wildlife Fund **.**

Case study 1: IWRM in Canada

Some countries are more proactive in their application of natural infrastructure solutions. Canada is one of them. There are many examples of IWRM initiatives of watersheds that list ecosystem protection among their IWRM goals. In fact, ecosystem protection was indicated as one of the IWRM goals for 14 out of the 35 examined watersheds which implement IWRM (Roy, Oborne&Venema, 2009). For example, IWRM-related goals in Saskatchewan are directed towards "understanding source water quality risks, and maximizing watershed protection through natural purification and other means to minimize contamination potential" (Roy, Oborne&Venema, 2009).One of the impressive examples is the use of natural infrastructure in order to tackle the eutrophication problem in Lake Winnipeg. It was proposed that reeds (which are known to be very efficient in phosphorus accumulation) should be harvested and processed, and that the obtained phosphorus be used for agricultural purposes and the reeds themselves for biofuel production.

Case study 2: San Jeronimo Basin, Baja Verapaz, Guatemala (Oliva, 2008).
The San Jeronimo River Basin is located in a mountain region surrounded by lavish tropical forest, blessed with rich biodiversity and water resources. The picturesque landscape and rich biodiversity hold great opportunities for the development of eco-tourism. However, there are several growing environmental concerns in the region, including deforestation, improper waste water treatment, and the overuse of water resources in the basin. These have all led to serious water problems, both in terms of quality (pollution of ground water/surface water) and quantity. It has been recognized that water is a critical resource, and that the preservation of forests is crucial to ensure its quality and availability.

IWRM actions. The San Jeronimo Basin Committee was established to ensure the sustainable use of the water resources in the basin. The Committee holds a common discussion forum for the representatives of all main water users, and addresses issues such as irrigation, aquaculture, hydroelectricity, tourism, domestic users, etc. Additionally, it fosters coordination and negotiation among all the stakeholders. In particular, the committee directs water management actions to preserve water resources and ecosystems in the river basin, thus making sure that the systems are properly maintained and that the water quality and quantity levels can improve.

Results. The establishment of the committee strengthened the coordination among all the stakeholders, and improved joint efforts to promote sustainable water use, thus helping to preserve the basin. The Association of Users of Irrigation of San Jeronimo consists of 800 users and is responsible for the reforestation of 30 hectares at the heart of Sierra de las Minas, which has been designated as a Biosphere Reserve. The committee's activities have resulted in an increased acceptance of social responsibilities by organizations, and have thus led to positive results in water conservation. This case study illustrates the specific goals of water and ecosystem conservation applied to resolve water issues in the basin; however, water infrastructure solutions—besides the conservation of basin resources themselves—pertain only to reforestation activities, financially supported by the Payments for Ecosystem services. There are no specific benefits mentioned.

Case study 3: Lake Rotorua, New Zealand (Roy,Oborne&Venema, 2009).
Intensive agricultural activities and the resulting nutrient discharges have contributed to the eutrophication of Lake Rotorua. To tackle this issue, various measures have been proposed, including natural infrastructure solutions, such as vegetative filter strips, constructed wetlands, and harvesting aquatic plants from farm streams and water courses.

9.4 Improving the mainstreaming natural infrastructure in IWRM

What is essential to realize is that viewing ecosystems as necessary infrastructure for water provision and regulation puts ecosystem protection in a positive light, not as a separate, often conflicting issue with water problems; it re-frames them as a solution to water problems. Measures which concentrate on the improvement of soil, forests, and wetlands—water's natural infrastructure—have been shown to effectively tackle the root problem of water crises. Enhancing ecosystem integrity and health is imperative for resolving water issues. In summary:

- There are very limited examples of managing ecosystems (natural infrastructure together with/without built infrastructure) to achieve water management goals. This may imply that currently, in the practical application of IWRM, ecosystems are not viewed as part of the solution to water problems, and are rather viewed as a constraint to development.
- IWRM has been criticized for its insufficient coverage of environmental aspects, and for being primarily focused on water allocation and pollution, not on ecosystem health.
- IWRM's focus is on managing the demand of water resources; however, it is not widely recognized that by applying natural infrastructure solutions, one can increase water supply and manage the supply side

What are the key obstacles to the wider application of natural infrastructure solutions? One of the major issues is the lack of awareness and understanding of the complexity of the water cycle and its dependence on biodiversity—a fact which underpins ecosystem services. It is generally recognized that water is essential for the proper functioning of ecosystems. However, rarely are explicit links made between healthy ecosystems and the maintenance of water quantity and quality.

Coates and Smith (2012) explain that one major obstacle to the wider application of natural infrastructure solutions in IWRM is that ecosystems are viewed as problems, not as assets to tackle water issues. The authors point out that the traditional approach to water management addresses water problems using engineered solutions, and regards the environment as an inevitable cost of development. Benefits produced by ecosystems are considered, but not incorporated into management options. This results in negative impacts on ecosystems and the ecosystems being "perceived in conflict with human needs". Therefore, they propose a new paradigm (Figure 9.1), where ecosystems (natural infrastructure) are managed together with a built infrastructure. Maintaining or even improving ecosystem health and resilience safeguards the provision of quality water of a sufficient quantity as well as other ecosystem services, including flood attenuation, soil erosion reduction, and the control of sedimentation. This in turn reduces the costs associated with built infrastructure and reduces the risks associated with climate change.

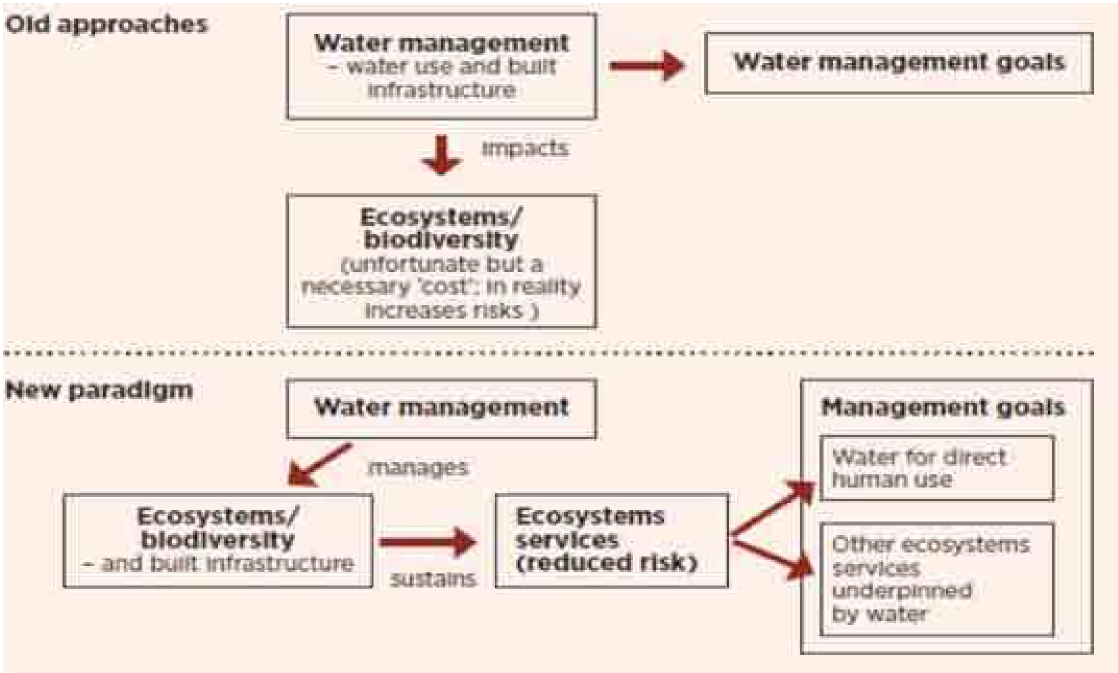


Figure 9.1: Changing paradigms for water management. Adopted from WWAP, 2012.

The authors describe a three step approach to ecosystem management: 1) setting water management goals for direct human use (i.e. improving water quality or the provision of sufficient amounts of drinking water); 2) examining how these can be achieved by enhancing/maintaining ecosystem services; and 3) considering all the ecosystem services impacted by the decision, in particular, evaluating the co-benefits provided by ecosystems and "examining trade-offs between them to determine desirable courses of action".

Taking full account of environmental assets might be challenging considering high priority issues such as critical water shortages, water pollution and sanitation problems. However, it is critical to understand that "user interests coincide with the environmental issues provides clear benefits as opposed to acting as constraints". Therefore, environmental assets and the potential trade-offs with other management options should be considered at the beginning of the IWRM planning process.

Many water experts have identified that there is a lack of global environmental data (WWAP, 2012), such as data on "the condition and extent of wetlands - an important gap considering their hydrological functions", monitoring data of nutrient cycling, data on sediment transfer and

depositions, and finally data on water quality (WWAP, 2012). Attempts to make relevant connections between biodiversity, ecosystems and people are improving, but they need to be scaled up even further. Another potential constraint to the adoption of ecosystem solutions is that they "offer less opportunity for corruption" (WWAP, 2012).
What facilitates the incorporation of natural infrastructure solutions? Significant progress in the implementation of natural infrastructure solutions in IWRM can be achieved by applying ecohydrological concepts and methodology.

9.5 Ecohydrology

There is a growing interest in the use of natural infrastructure solutions around the world. A sub-discipline of hydrology, ecohydrology, was developed to provide sound science to water managers and decision-makers in response to the shortcomings of engineering solutions, which don't account for ecological processes. The primary goal of water management based on ecohydrology is "the enhancement of a catchment resilience leading to improvements in water resources (quality and quantity) and ecosystem status (biodiversity and ecosystem health) while meeting the needs of water users" (Wagner *et al.*, 2009). Ecohydrology develops solutions to specific water issues by combining knowledge of ecological processes and hydrological processes. Growing interest in the field is concerned with providing more cost-efficient, decentralized solutions based on natural systems that perform diverse services such as water purification, nutrient recycling, carbon sequestration, etc. Ecohydrology in general requires knowledge of the temporal/spatial patterns of catchment-scale water dynamics, determined by the four fundamental components: climate, geomorphology, plant cover/biota dynamics and anthropogenic modifications. An ecohydrological approach to water management aims not only at reducing and eliminating water contaminants, but also at the "amelioration of the effectiveness of potential tools to manage the dynamics of excess nutrients, pollutants, mineral and organic matter in the landscape". This can be done by reducing human impact and regulating the aquatic and terrestrial biota in the catchment. Ecohydrological solutions include (Wagner *et al.*, 2009):

- Increasing watershed water retention through reforestation
- Enhancing in-stream retention of water sediments and nutrients through river re-naturalisation and wetland restoration,
- Amplification of biogeochemical cycles such as denitrification through wetland inundation

Water quality maintenance can be significantly reduced by applying measures that are based on biotic processes, which can facilitate self-purification in water bodies. Ecohydrology Centres have been established under the auspices of UNESCO in Indonesia, Ethiopia and Poland. There are numerous projects in the world demonstrating the effectiveness and economic feasibility of ecohydrological solutions together or without built infrastructure solutions. Two of them are the Pilica River (Poland) and Lake Naivasha Basin demonstration projects, developed within the framework of the International Hydrology Programme of UNESCO.

The Pilica River Project
The Sulejow Reservoir in the Pilica River basin was constructed to supply water for the city of Lodz, a popular destination for people pursuing recreational activities (Wagner *et al.*, 2009). This project demonstrates appropriate methodology for the implementation of ecohydrological approaches into IWRM. It consists of four steps:

1) Monitoring threats
2) Assessing cause-effect relationships
3) Developing ecohydrological methods
4) Developing system solutions.

Sixty four percent of the catchment is used for agricultural purposes (Wagner *et al.*, 2009). First, in the monitoring phase, appearance, intensity and/or spatial dynamics, and the risks to society were examined. The water issues identified at the Pilica River were three-fold: water contamination with dioxin-like pollutants, the risk of eutrophication, and sedimentation. Sources of pollution included stormwater runoff from the town (which had an inefficient treatment plant) and runoff from the agricultural fields. To reverse the siltation of the lake and reduce the contamination of the lake water, researchers developed and applied ecohydrological solutions. The methods used to manage toxic algal blooms were focused on reducing pollution loads into the reservoir and controlling the levels of cyanobacteria within the reservoir.
The proposed ecohydrological solutions included:

- Diversion of the highest pollutant load into floodplain areas for nutrient retention through sedimentation and assimilation in the vegetation (willow patches)
- Hydro biomanipulation –regulating water level reducing juvenile fish density- to remove the predation pressure on the zooplankton consuming algae
- Optimization of denitrification process through regulating the water retention time

Lake Naivasha Basin Project
In the last two decades, Lake Naivasha has witnessed an almost utter destruction of its ecosystem services. The destruction of the second-largest lake in Kenya is attributable to the introduction of various alien species and to the effects of the horticultural industry. The resulting increase in sediment and nutrient inputs has led to many negative consequences, including biodiversity loss and habitat degradation. In response to this problem, there have been calls for considering it on a basin, rather than lake, level, as well as changing the mindset of problem solvers from “water” to the “water cycle”. Thus, Ecohydrology was suggested as a possible remedy to resolve these needs, with the first step being the recreation of a papyrus buffer zone around the lake’s shoreline.

9.6 Payments for Ecosystem Services

Importantly, the adoption of economic mechanisms, such as payment for ecosystem services, can greatly contribute to the financial sustainability of ecosystem protection measures. Such financial support can stem from combining the goods and services jointly produced by the watershed. These efforts allow for a greater degree of watershed preservation than hydrological services alone. They additionally allow for new financing opportunities for ecosystem services which may be complementary to the natural water supply/purification purposes by watersheds. These include soil conservation, sedimentation control, fisheries protection, carbon sequestration, biodiversity conservation, recreation, tourism, and cultural and aesthetic enhancements. For example, Ecuador’s National Biodiversity Policy encourages beneficiaries to pay for a variety of environmental services, including the provision of water, on public and private lands (Postel & Thompson, 2005).

9.7 Recommendations

Natural infrastructure solutions, along with built infrastructure options, should be incorporated into cost/benefit valuation analyses, especially when it comes to considering projects such as dams, storage, irrigation, and drainage. Recommendations for improving current levels of consideration for environmental aspects in IWRM might include:

- Considering environmental water issues early in IWRM planning and focusing on environmental water issues, not only institutions or instruments (UNEP, 2005)

- Making a cause-effect assessment of key environmental priority issues related to the appropriate approach in the specific context (UNEP, 2005)
- Strengthening of multi-stakeholder national IWRM networks and partnerships with focus on awareness-raising and advocacy on environmental aspects in IWRM
- Defining environmental aspects in national IWRM:
  -assessing ecosystem value with scarce information
  -major environmental water issues
- Viewing water resources as more than just river discharge and water quality, especially when it comes to rain-fed agriculture (this would have important implications for food security and poverty alleviation). Many assets of ecosystems are rarely accounted for. Wetlands ecosystems also provide goods and services in a form of crops, fish, timber which are vital for the poor.
- Providing politically convincing and tangible examples of ecosystem development and poverty alleviation potential in ecosystem goods and services
- Generating self-sustainable funding for ecosystem protection such as payments for ecosystem services .

The Global Water Partnership IWRM toolbox might be enhanced by include ecohydrological solutions and support measures strengthened by promoting collaborative learning processes among different stakeholders. For the CBD, there are opportunities to promote reporting on natural infrastructure solutions, including with regards to IWRM, in national reports, as well as further collaboration with the GWP and other partners for information sharing, establishment of common goals and further mainstreaming of natural infrastructure solutions.

Natural infrastructure solutions are not a silver bullet for all water problems: their effectiveness varies depending on constrains of the natural environment as well as social, economical and political context. Proponents of the approach need to bear this in mind, as well as the overall difficulties in implementing IWRM, so as not to promise more than can be realistically delivered.

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